RI FINAL

VOL I. Sec. 19-39
APPENDICES AND FIGURES
AND TABLES

K.8
6/88

SECTION 19 - SITE 12, AREA 14 IMPOUNDMENT

19.1 Site Description

Site 12 is located within Area 14, which is a manufacturing area on the south side of the lake (See Figure 19-1). Munitions loading activities occurred in Area 14 in the past. Many of the buildings have been abandoned or demolished, but a few industries continue to operate at Area 14. The area in the vicinity of Site 12 is currently occupied by Diagraph-Bradley for the manufacture of printing inks, stencils, stencil boards, and marking pens.

Historic aerial photographs indicate apparent impoundment activity in Area 14 east of the presently occupied buildings in 1943. By 1951, the impoundment had been taken out of service and the area was partially vegetated. The aerial photographs show what appears to be an aboveground tank situated in the middle of the former impoundment area. The access road is still visible in the 1951 photographs. Photographs from 1960 still show the tank, with the surrounding area completely grown over with vegetation. The access road appears unused in these photographs. The next set of aerial photographs represent 1965 conditions and indicate that the tank has been removed.

The site is currently a circular dry impoundment with a diameter of approximately 100 feet. The interior of the impoundment is presently overgrown with trees with trunk diameters of 8 to 10 inches. The impoundment walls are about 6 feet high and the north wall was breached between 1960 and 1965, according to the air photos, to allow drainage to flow from the impoundment to an adjoining field. Several black oily pools are evident in and around the basin. Other bare patches of black sediment and tars are located around the basin floor.

19.2 Site Investigations

19.2.1 Phase I Site Investigations:

One composite soil sample (0-1 ft depth) and one composite sediment sample (0-1 ft depth) were collected. The sediment sample was later resampled for full priority pollutant analysis. The scheduled water analysis was canceled because the site was dry at the time of sampling.

19.2.2 Phase II Site Investigations:

No samples were collected in Phase II.

19.3 Analytical Results (See Appendix I, Page 12)

TOC levels in soils and sediments ranged from 12,039 to 16,673 mg/kg, while TKN concentrations of 369 to 2,267 mg/kg were detected. The results of the FID screening were 16,934 ug/kg, resulting in the collection of an additional sample for full priority pollutant analysis. The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present. N-nitrosodiphenylamine was detected in the sediment at 2,174 ug/kg wet weight. Small concentrations (less than 1 mg/kg) of other base/neutral extractable compounds (phenanthrene, pyrene, fluorene and anthracene) were quantified in the sediment but were present below the detection limits.

19.4 Environmental Effects

19.4.1 Qualitative Assessment

This site was chosen for investigation based on a history of munitions activity at this location. Historical aerial photography reveals

that the site was utilized as an impoundment as early as 1943.

Concentrations of TOC, TKN, and FID were detected in soils and sediments, but were consistent with those of the control sites. N-nitrosodiphenylamine was detected in the sediment sample, at a level higher than at any other site at the Refuge. N-nitrosodiphenylamine (NDA) has produced a carcinogenic response in laboratory tests with animals, thus, it will be considered as the site indicator contaminant in the quantitative assessment for humans and wildlife. As a conservative measure, the level of NDA detected in site sediment (2,174 ug/kg) will be considered as two times this level (4,348 ug/kg) due to deficiencies in the Phase I laboratory analysis.

In addition, trace base/neutral extractable compounds below 1 mg/kg detected in sediments included phenanthrene, pyrene, fluorene, and anthracene.

19.4.2 Quantitative Assessment

Humans

During the Phase I investigation, the only elevated contaminant of concern was N-nitrosodiphenylamine in the impoundment sediment, at a level of 2,174 ug/kg. Since the impoundment is currently overgrown with trees and is also surrounded by a berm, it is unlikely that a complete pathway exists for any human exposures to this contaminant source. However, in order to provide some framework for the upper bound risks that could be posed by chronic exposure of a site visitor to residues of this magnitude, the following scenario was analyzed.

It is assumed that a human visitor to the site might ingest an average of 100 mg of impoundment sediments as a result of direct contact

with the material. It is also assumed that the sediment contains 4,348 ug/kg of N-nitrosodiphenylamine, twice the level found in the single sediment sample analyzed, in order to provide a conservative upper bound on the residue level in the absence of more sample data. Exposure at this level would result in an average intake of 0.435 ug N-nitrosodiphenylamine per visit. Assuming a 70 kg adult such as a site inspector or Refuge employee might visit this site roughly three times yearly over a lifetime of 70 years, and using the EPA carcinogenicity potency factor of 0.005 $(mg/kg/day)^{-1}$ (Exhibit A), an excess lifetime risk of cancer of 2.5×10^{-10} is estimated. Given the upper bound assumptions employed in this assessment, and the consideration that "acceptable" risk levels are construed as levels below 10^{-4} (USEPA 540/1-85/060 1986), this represents a negligible level of risk.

Wildlife

The detection of N-nitrosodiphenylamine in a soil sample analyzed in the Phase I survey of this site also presents a mechanism for exposure for terrestrial wildlife via the direct contact route. The levels of exposure would be greatest amongst small mammals as a result of inadvertent ingestion and inhalation of contaminated soil residues and dust during daily burrowing, feeding and grooming. Thus, the risks of direct contact of these species to site nitrosamines residues were assessed. The risks to larger and/or less sensitive species, or to those which have less contact with soil residues would be lower. A search of on-line data bases (Pollution Abstracts, Biosis Previews, NTIS, HSDB) did not identify published studies on the effects of N-nitrosodiphenylamine on pertinent wildlife species. Therefore, tests with surrogate species (i.e. laboratory rodents) are used in the assessment below.

Using a breathing rate of 0.006 m3/hour for an active 30 g mouse (USEPA ECAO-CIN-477, 1985), and creation of a 10 mg/m3 dust containing 4,348 ug/kg N-nitrosodiphenylamine during one hour of daily burrowing, a daily chronic inhalation exposure of 0.0087 ug/kg/day is obtained. Exposures via the ingestion route due to consumption of contaminants in soil or vegetation are estimated at 0.041 mg/kg/day using an ingestion rate of 5% body weight or 1.5 g food/day (10% soil, and 90% vegetation containing 1% of the soil concentration). These exposures are calculated assuming levels in soil at twice the measured concentration of contaminant. Thus a total daily exposure rate of 0.041 mg/kg/day is estimated for small burrowing animals from inhalation and ingestion at this site. Using the EPA carcinogenicity unit risk factor of 0.005 $(mg/kg/day)^{-1}$ for N-nitrosodiphenylamine (Exhibit A), developed on the basis of rat studies, the total estimated daily chronic exposure would result in a risk level of 2.0×10^{-4}. The significance of this exposure is discussed below.

As discussed by Newell et al.(1987), concerns regarding the effects of cancer on wild populations are largely unknown, and risk levels of concern to humans are not directly transferable to wildlife. Many other factors come into play when addressing whether a wildlife population can maintain itself (i.e. survival to reproductive age, competition, weather, disease, predation, etc), and the effect of cancer, generally forming later in an exposed organism's lifetime, might thus be very small. On this basis, Newell et al. (1987) chose a risk level of 10^{-2} as a level of acceptable carcinogenic risk for wildlife, with the acknowledgement that more study is needed to justify this choice. Using this rationale, it is concluded that wildlife exposure to site residues of N-nitrosodiphenylamine are well below the level which could result in a carcinogenic response.

Additional review of the literature on the effects of N-nitrosodiphenylamine to wildlife is presented in the Toxicological Profile for N-nitrosodiphenylamine (ATSDR, 1987). Using the subchronic no observed adverse effect level of 150 mg/kg/day for oral exposure of laboratory rats to N-nitrosodiphenylamine derived in this review, and a safety factor of 10, a wildlife ADI of 15 mg/kg/day is derived. The estimated exposure of 0.047 mg/kg/day for burrowing rodents at this site is well within this acceptable intake level.

19.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a history of munitions activity on the site, aerial photography, site inspection, and sample analyses. A visual survey of the site included black oily pools in and around the basin. Black sediment and tars were also observed on the basin floor.

Chemical residue information consisted of analytical results for surface soil samples. The soil sample represented only the top one foot of soil; deeper soil borings and ground water monitoring were not conducted. Since contamination of the site occurred possibly through surface spillage of waste, soil and sediment contamination would most likely be found at the surface. Since there is no evidence to suggest that the surrounding soil or sediment has been excavated or otherwise disturbed, these samples should adequately represent the conditions of the site.

The quantitative risk assessment was performed under a worst case chronic exposure scenario of lifetime exposures and considered only the single data point available for a soil sample in which

N-nitrosodiphenylamine was detected in the Phase I survey. This result is only qualitatively reliable due to insufficient QA/QC supporting the analysis. In addition, the residue concentration detected was doubled in the risk calculation to provide a more conservative outcome. This worst case approach resulted in risk levels that provide an ample margin of safety for protection of potential human and wildlife receptors under the conditions assumed.

19.5 Preliminary Remedial Alternatives

Preliminary Phase I screening results discussed in the previous section indicated that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore this site was not included in the Phase II investigations. There will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

19.6 Conclusions and Recommendations

It can be concluded that the Area 14 Impoundment site does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

SECTION 20 - SITE 13, AREA 14 CHANGE HOUSE

20.1 Site Description

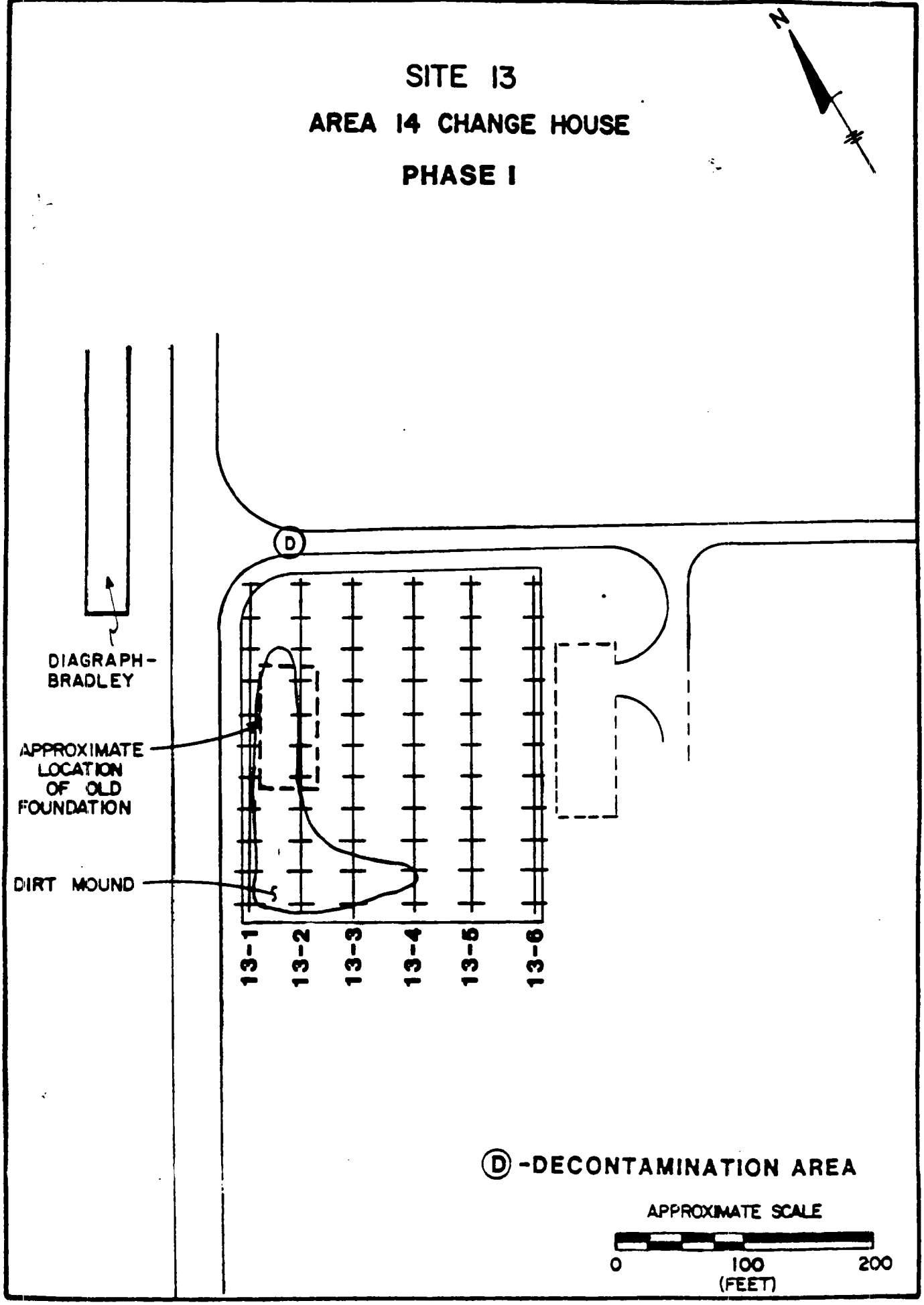
Site 13 is located southeast of the active Diagraph-Bradley buildings in Area 14 (See Figure 20-1). Further information on Area 14 can be found in Section 19.1. Site 13 consists of the site of a building which was demolished sometime between 1971 and 1980, according to aerial photographs from those years. The site is presently an open field covered with tall grass. Formerly, it was the site of a "Change House" where workers changed their clothing after working in the adjacent bomb-loading buildings. At one time a company named CTI (possibly Chemicals and Technology, Inc.) manufactured explosives and other chemicals in this building, according to the Refuge Manager. Other industries may also have occupied this building. The change building was located across from the bomb-loading building on a plot of land just southeast of the intersection of two roads on the north edge of a big dirt mound. The concrete floor of the Change House is under this mound, according to the Refuge Manager. Aerial photos show another building (no longer present) further east of the corner; field inspection revealed several half-inch reinforcing rods imbedded in concrete near the corners of the building.

20.2 Site Investigations

20.2.1 Phase I Site Investigations

A magnetometer and electromagnetic terrain conductivity survey was conducted over a 250 ft x 200 ft area, with grid spacings of 25 ft centers. The results of these surveys are shown on Figures 20-2 and 20-3. No magnetic anomalies of any significance were noted, indicating that there is no buried metallic debris at the site.

FIGURE 20-1



O'BRIEN & GERE ENGINEERS INC

FIGURE 20-2

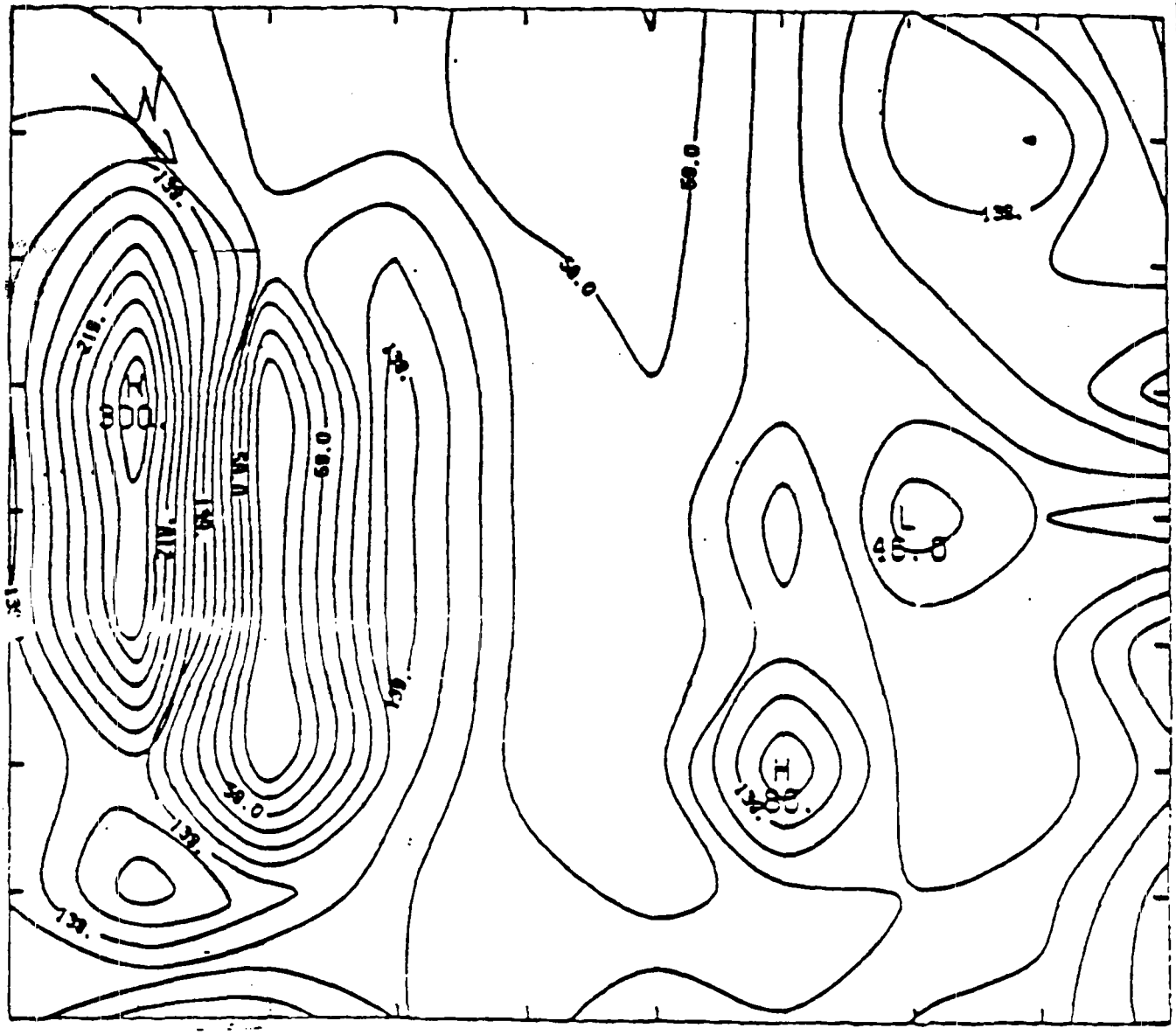
SITE 13
MAGNETOMETER SURVEY

O'BRIEN & GERE ENGINEERS INC

FIGURE 20-3

SITE 13

ELECTROMAGNETIC SURVEY



Six composite soil samples (0-1 ft depth) were collected along north-south transect lines (Figure 20-1) and screened for priority pollutants, metals, cyanide, indicators and explosives.

20.2.2 Phase II Site Investigations

No additional sampling was done in Phase II.

20.3 Analytical Results (See Appendix I, page 13)

The geophysical surveys did not indicate that major buried articles are present. The soil concentrations were consistent with those detected at the control sites, although these concentrations were estimated for screening purposes only (see Exhibit B). No organic compounds were detected with the exception of delta-BHC in one soil sample (69 ug/kg), which was slightly over the detection level.

20.4 Environmental Effects

20.4.1 Qualitative Assessment

This site was chosen for investigation based on verbal accounts regarding a history of munitions activity at this location. There was, however, no history of the disposal of wastes and the historical aerial photography review as well as the geophysical surveys did not reveal the existence of a waste disposal area. The absence of a waste disposal area was further supported by the results of the Phase I sampling. On this basis, it can be concluded that there is no "source" of waste materials for on-site exposures or for migration to off-site locations.

Because there is no established waste source at this location, it is not possible to have a "complete" exposure scenario. Therefore, on the

basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

20.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

20.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was the verbal accounts of activities on the site, aerial photography, and site inspection. The verbal accounts suggested that the area could have wastes on it, although the use of the area for waste disposal is not indicated.

Chemical residue information consisted of analytical results on surface soil samples. The soils sampled represented only the top one foot of soil; deeper soil borings and ground water monitoring were not conducted. It appears that the ground was disturbed during the time that the change house was demolished. The surface soils could have been moved around or dirt hauled in to cover the building foundation. Surface sampling would be adequate for determining that there are no contaminants on the present soil surface, but it would not be adequate for identifying munitions contaminants present on the soil surface (now possibly buried) that were exposed at the time the change house was in use. Therefore no conclusions regarding subsurface conditions could be drawn exclusively on the basis of the residue information. However, the

geophysical surveys did not suggest the presence of unexplained subsurface metallic anomalies.

It can be concluded that the data generated are adequate when considered in light of the fact that there was no known history of waste disposal at this location. Residues related to the changing of uniforms soiled during munitions loading activities would have settled on the surface soils of the area. The surface soil sampling program would therefore be an adequate means of locating and identifying munitions related chemical residues.

20.5 Preliminary Remedial Alternatives

Preliminary Phase I screening results discussed in the previous section indicated that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore this site was not included in the Phase II investigations. There will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

20.6 Conclusions and Recommendations

It can be concluded that the Change House site does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

SECTION 21 - SITE 14, SOLVENT STORAGE

21.1 Site Description

Further information on Area 14 and Diagraph-Bradley is presented in Section 19.1. Linseed oil and various solvents are handled in bulk and in drums. Some of the bulk solvents noted were: T25 Xylene, T8 Diacetone Alcohol, T9 Diethylene Glycol, and T18 Methyl Cellosolve. Several compressed gas cylinders are also present. At least two drum storage areas containing 50 to 200 drums were also noted. Spill containment facilities are minimal. Site 14 is a drainage ditch adjacent to the active manufacturing operations of Diagraph-Bradley (See Figure 21-1). The ditch receives runoff from a manufacturing area where solvents are handled in bulk and in drums. The ditch runs north parallel to the road west of the buildings. Process water from the Diagraph-Bradley buildings enters this ditch from a standpipe.

21.2 Site Investigations

21.2.1 Phase I Site Investigations

Two composite water samples and two composite sediments (0-1 ft depth) were collected. One sediment composite was resampled for full priority pollutant analysis.

21.2.2 Phase II Site Investigations

One composite water sample and one composite sediment sample were collected by the drum storage area (See Figure 21-2). The Phase II samples were analyzed for purgeable and base/neutral/ acid extractable organics.

FIGURE 21-1

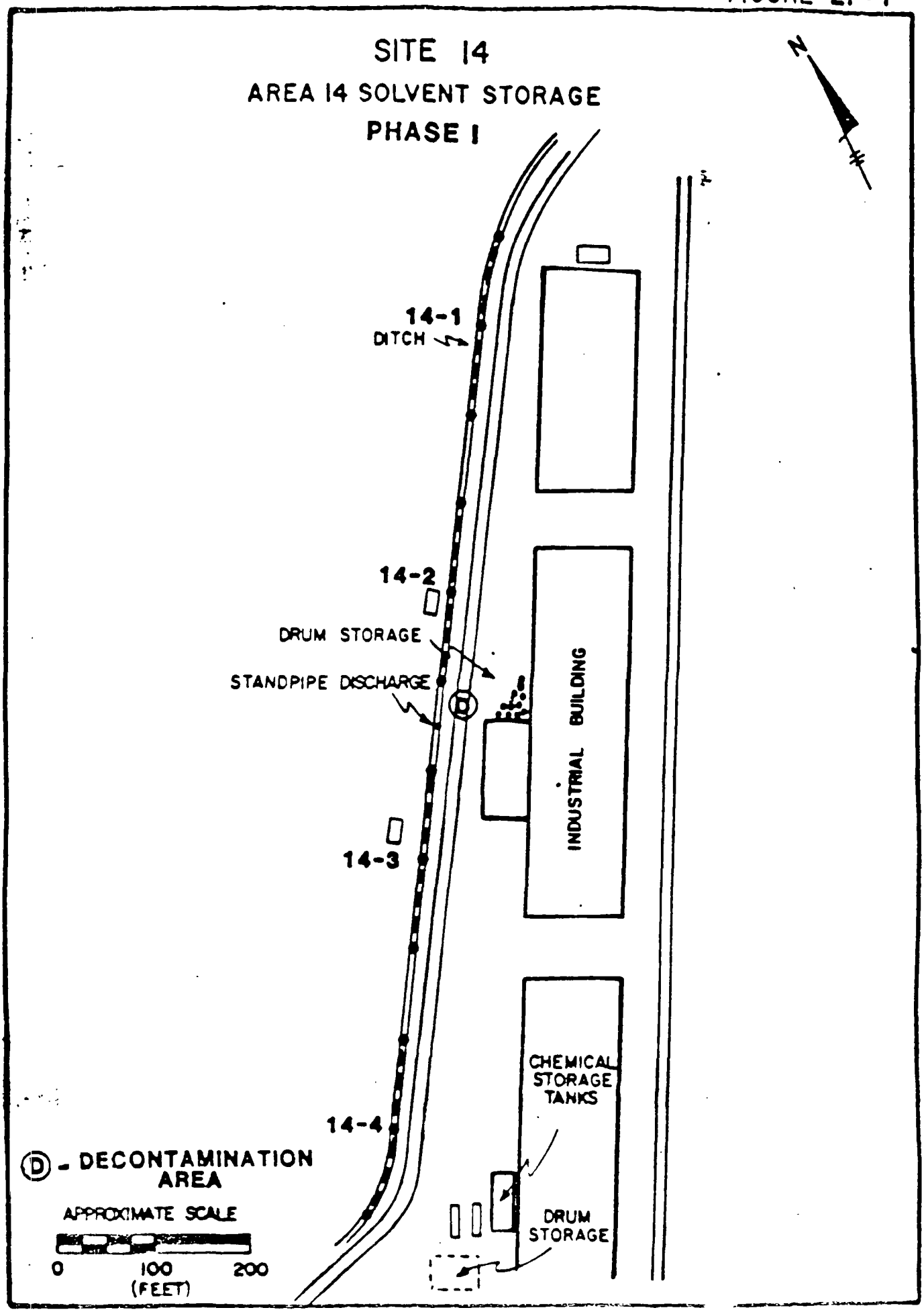
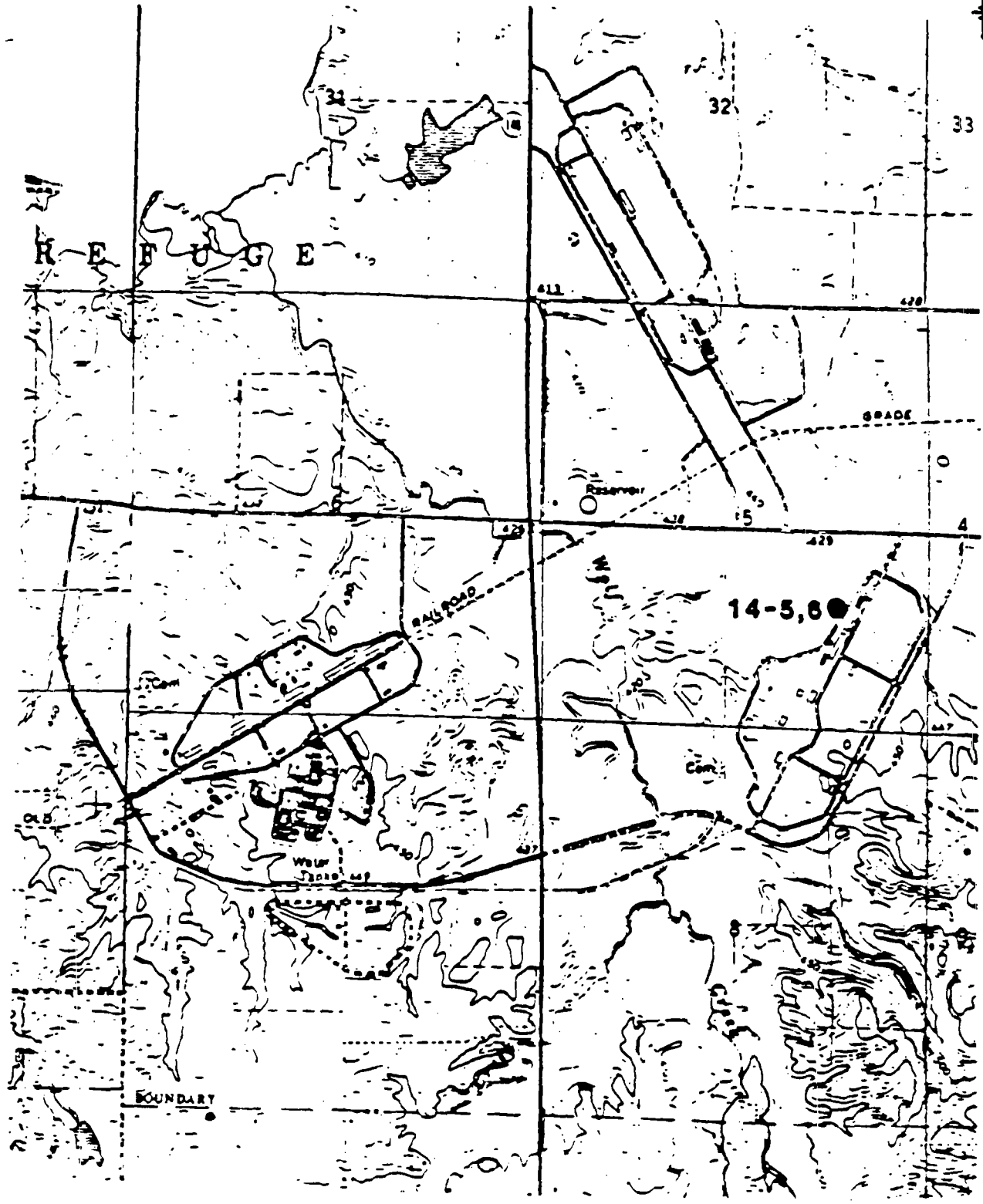

SITE 14
AREA 14 SOLVENT STORAGE
PHASE I
N
14-1
DITCH
14-2
DRUM STORAGE
STANDPIPE DISCHARGE
D
INDUSTRIAL BUILDING
14-3
CHEMICAL STORAGE TANKS
14-4
DRUM STORAGE
D - DECONTAMINATION AREA
APPROXIMATE SCALE
0
100
200
(FEET)

FIGURE 21-2

SITE 14
SAMPLING LOCATIONS
PHASE II



21.3 Analytical Results (See Appendix I, page 14)

21.3.1 Phase I Analytical Results

The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported below are thus estimated values and some compounds which were not detected may in fact be present.

Chloroform was detected in the water samples at 11 and 43 ug/L, which was above the ambient water quality criteria for human health of 0.19 ug/L. Iron and manganese levels in water (600 and 180 ug/L) were above the Federal MCLs and Illinois Public Water Supply Standards, but only the manganese concentration was above the Illinois General Use Water Supply Standard of 150 ug/L. Iron and manganese standards are based on concerns for taste and color, such that the levels detected are not indicative of any site risks. Bromodichloromethane was also detected at a concentration of 5.0 ug/L. Subsequent to an FID screening of 36,704 ug/kg, one of the sediment samples was selected for CLP organics. This sample contained trace organics below the detection limits or at levels consistent with those detected at the control sites or in the QA/QC blanks. N-nitrosodimethylamine (95 ug/kg wet weight), methylene chloride (657 ug/kg wet weight), and acetone (188 ug/kg wet weight) were among the compounds which were detected.

21.3.2 Phase II Analytical Results

Analytical results for the water sampled showed 123 ug/L chloroform, as well as traces of acetone (43 ug/L) and bromodichloromethane (23 ug/L). Methylene chloride was detected in the water (3-15 ug/L) but

also in the QA/QC blank. The spike recoveries for the CLP analysis in water were outside of QC limits. All parameters were below the Federal MCLs and Illinois State Public Water Supply Standards, but chloroform exceeded the AWQC for human health of 0.19 ug/L. The sediment contained 4-methylphenol (273 ug/kg), bis-(2-ethylhexyl)phthalate (270 ug/kg), di-n-butyl phthalate (1680 ug/kg), acetone (6480 ug/kg) and methylene chloride (676 ug/kg). No other organics were detected in the sediments; however the volatiles were analyzed outside the holding time and the semi-volatiles were outside the QC limits for spike recoveries. All other parameters were detected at concentrations similar to those detected at the control sites.

21.4 Environmental Effects

21.4.1 Qualitative Assessment

21.4.1.1 Source Evaluation

As discussed in the preceding sections, the Solvent Storage Site is a ditch which receives effluents from an active manufacturing facility. Its upgradient proximity to Crab Orchard Lake creates the potential for offsite contaminant transport to the lake. Several compounds were detected in ditch water and sediment including N-nitrosodimethylamine, methylene chloride, acetone, 4-methyl- phenol, phthalate esters, chloroform, and bromodichloromethane. Of the contaminants in water, only chloroform exceeded Federal water quality criteria.

The physicochemical and toxicological properties of site contaminants of concern are summarized in Exhibit A. The detection of N-nitrosodimethylamine in ditch sediments is significant due to concerns that it might be carcinogenic in humans. However, the residues detected

are of the same order of magnitude as detected at the control sites, and are not supported by QA/QC data. For these reasons, N-nitrosodimethylamine will not be considered further in this risk assessment. Di-n-butylphthalate was detected in sediments but will not be used as a site indicator parameter since it was also present as a laboratory contaminant. In addition, due to its small size and intermittent dry periods, the ditch does not apparently support significant population of aquatic organisms which might be exposed to phthalates or other site contaminants. The volatility of chloroform and bromodichloromethane compounds from water is high and it is highly unlikely that these residues would persist for a sufficiently long period of time to be transported offsite (Callahan et. al., 1979). For these reasons, chloroform and bromodichloromethane detected in water at this site were not chosen as site indicator compounds. In contrast, methylene chloride was detected in both water and sediment matrices and thus, although it may be highly volatile from water, its presence in sediments will result in its persistence in the stream. It should be noted that methylene chloride was detected at very low levels in the water column as well as in the water blanks; it was not detected in the QA/QC blank for sediment.

Due to persistence, relative residue levels, and concerns for toxicity, methylene chloride was chosen as the site indicator contaminant. Methylene chloride is of concern due to its ability to induce damage to several organs in animal studies, mutagenicity, and evidence suggesting that it may be carcinogenic in animals. Available data show a low degree of toxicity in aquatic organisms. Although methylene chloride may also be formed during the chlorination of plant effluents, it is also a major industrial solvent. Its presence in the ditch sediments at this site in

relatively high concentrations suggests that it may have been discharged as a waste solvent.

21.4.1.2 Transport Route Evaluation

The site indicator contaminants have been detected only in the sediments of the drainage ditch. In view of the high water solubility of methylene chloride, residues in water might be expected. However, methylene chloride is also very volatile from water and may evaporate immediately under site conditions after discharge or desorption from the sediment.

a) Air: Due to the high volatility of methylene chloride, the airborne exposure route is functional.

b) Direct Contact: Due to the presence of sediments containing site contaminants in the ditch, the direct contact pathway is considered functional.

c) Surface Water: Even though site indicator residues were not detected in the water column in significant levels, their presence in sediments creates a functional pathway for offsite transport via surface runoff.

21.4.1.3 Receptor Evaluation

Human

Given the industrial nature of the site and non-populated downgradient land, the human receptor population will be limited to facility employees, site intruders, and, potentially, occasional hikers in downgradient areas.

a) Air Route: Since methylene chloride residues may volatilize from the ditch sediments and water, it is possible that site employees or intruders in the immediate vicinity of the plant may be exposed to methylene chloride vapor.

b) Direct Contact: Given the size of the manufacturing facility and the probable volumes of aqueous discharge, standing water will probably almost always be present in the ditch. Therefore, the likelihood that human receptors will come into direct contact with contaminated sediments is virtually nil, and this route is judged to be incomplete for human exposure.

c) Surface Water Route: Due to the rapid volatility of the site contaminant from water, human exposure to site contaminants transported off site are not probable, and this route will not be considered further.

Wildlife:

The most likely wildlife receptors to be exposed to site indicator residues include terrestrial wildlife in the immediate area of the source discharge, and downstream aquatic organisms which may inhabit the drainage system. Due to the small size of the ditch, it is subject to dry periods and therefore is not likely to support a significant aquatic population.

a) Air Route: As with human exposure by this route, any terrestrial wildlife in the immediate vicinity of the plant discharge may be exposed to vapors of methylene chloride. The lack of exposed contaminated dusts will minimize inhalation exposures to dust-bound contaminants.

b) Direct Contact: Benthic and bottom-feeding aquatic organisms may be exposed to site contaminants in the sediments of the drainage system downstream if sediments are transported offsite by surface runoff.

c) Surface Water: Since site indicator contaminants were not found in the water of the ditch, wildlife exposures by this mechanism will not occur.

21.4.2 Quantitative Assessment

21.4.2.1 Estimates of Release and Exposure

a) Air Route: The qualitative assessment determined that the air exposure route was potentially complete for humans and wildlife in the immediate vicinity of the source discharge due to the volatility of methylene chloride detected in sediments. Methylene chloride residues detected in the water column of the ditch were low, approximately 15 ug/L. Without data on the total volume of contaminated water at this location, an estimation of the steady state air concentration of methylene chloride is not possible. It is most likely that volatilized material will be immediately diluted to non-detectable levels and carried offsite by wind.

b) Direct Contact: The qualitative assessment determined that the direct contact route was complete for benthic and bottom-feeding aquatic organisms living downgradient in the site drainage system. These organisms may inadvertently ingest site contaminant while feeding within the sediment. Data on this ingestion rate were not located. For the purpose of this assessment, it is assumed that a benthic organism (i.e. annelid, insect larva) ingests 1 mg sediment

per g body weight. This would result in a daily intake of 0.7 ug/kg methylene chloride; the significance of this exposure is discussed in the quantitative assessment.

c) Surface Water: The surface water exposure route is complete for aquatic organisms inhabiting the drainage ditch. Methylene chloride was detected at 15 ug/L, these site residues will dissipate rapidly from the water by evaporation, unless continuously replenished by plant discharges. The quantitative assessment will consider the potential risk from these residues.

21.4.2.2 Quantitative Risk Assessment

Humans:

The qualitative assessment determined that the only complete exposure route for humans at this site would be acute or sub-acute inhalation exposure of plant workers to methylene chloride at levels which in all likelihood are below the limit of analytical detection. The health effects of methylene chloride are produced by chronic exposures and the acute toxicity of this compound is quite low. Therefore, it is concluded that short-term, acute exposure to methylene chloride residues which may exist at this site pose a negligible human risk.

Wildlife:

As reasoned above, exposure of terrestrial wildlife to airborne residues of methylene chloride poses a negligible risk due to the low levels of contaminant present. Aquatic organisms dwelling in the ditch downgradient of the source discharge, if any, may be exposed to concentrations of methylene chloride on the order of 15 ug/L,

without accounting for dilution of this contaminant in downstream areas where the ditch is somewhat larger. This concentration poses no concern for acute toxicity, although data on the chronic effects of methylene chloride are lacking. Downstream dissipation of residues would rapidly reduce chronic risks, if any, from methylene chloride exposure.

21.4.3 Analysis of Uncertainties

The principal areas of uncertainty are whether there is the potential for a viable benthic community in the drainage system in the absence of potential toxicants, and the potential low level, long-term effects of methylene chloride.

21.5 Preliminary Remedial Alternatives

The contaminants detected in the site investigations included volatile and semi-volatile organics. Methylene chloride was present in both the water and sediments in the ditch, making this contaminant the most persistent. However, the levels in water were not considered to pose a risk to humans or aquatic organisms (if any may inhabit the ditch) in or in the vicinity of the ditch. Due to the detection of some volatile organics in water at levels above the AWQC, and the proximity of active manufacturing areas, periodic monitoring of ditch waters might be conducted. Attachment 1 details a Monitoring Program which might be implemented. The monitoring activities together with improved housekeeping practices in the area could provide adequate protection of the environment and humans at this site. No further evaluation of remedial alternatives will be considered.

21.6 Conclusions and Recommendations

It can be concluded that the Solvent Storage ditch does not pose a risk to potential animal or human receptors, although improved maintenance practices in the area might be observed. The sediments and water were found to contain methylene chloride but not at levels which would pose a threat to exposed populations. This site will not be considered further in the FS. A monitoring plan for volatiles and semivolatiles in water is included as Attachment 1.

SECTION 22 - SITE 15, AREA 7 PLATING POND

22.1 Site Description

Site 15 is a pond located within a wooded rise to the south of Area 7 (See Figures 22-1 and 23-1). The pond reportedly received plating wastewater from previous Olin operations at the site. An inlet pipe is located on the northern side of the pond. There does not appear to be any outlet. The plating pond has dimensions of approximately 50 feet x 30 feet, and is bermed to about 5 feet above the water level. The water in the pond was estimated to be about 4 feet deep at the time of the inspection. Frogs were observed in the pond.

22.2 Site Investigations

22.2.1 Phase I Site Investigations

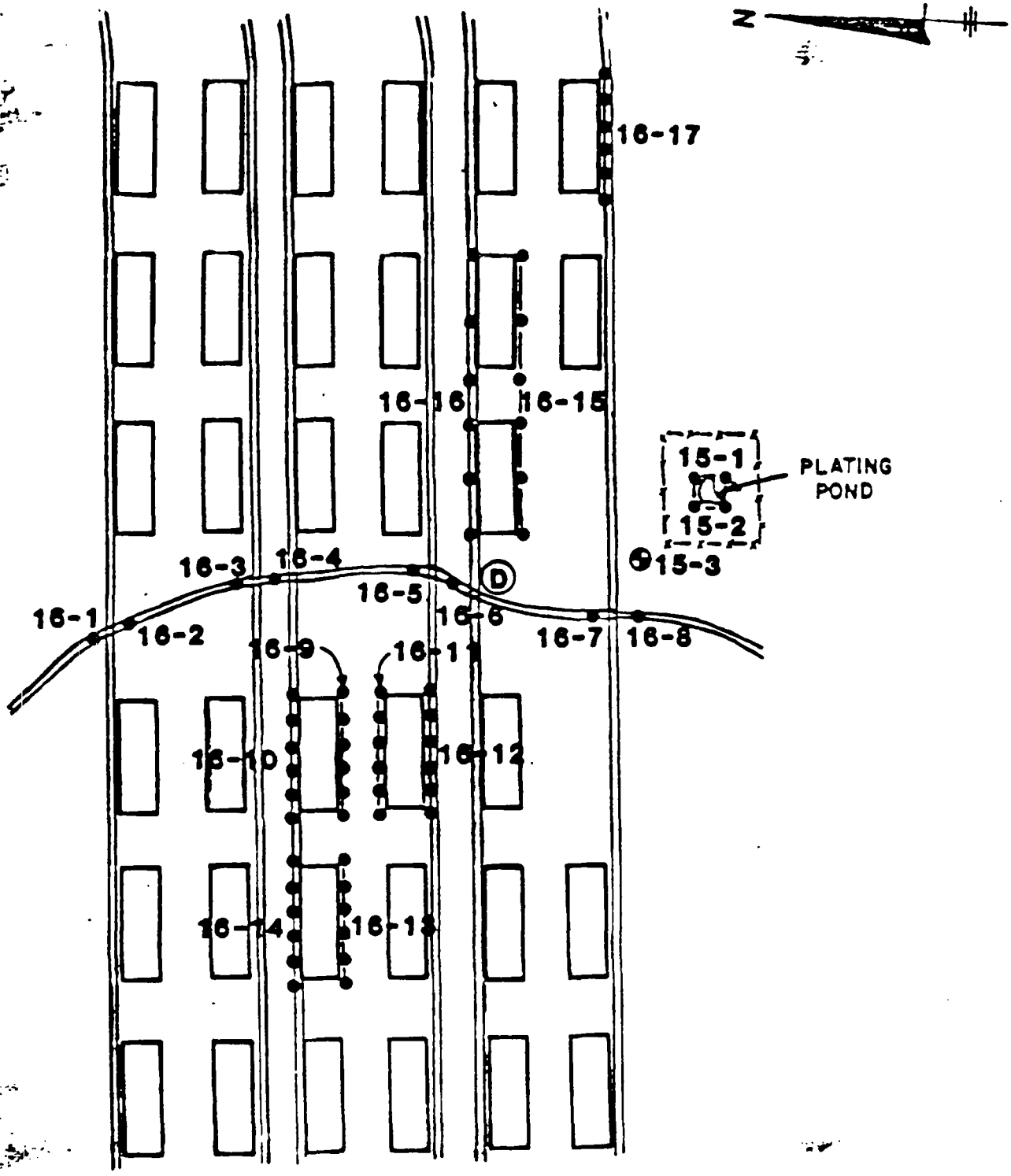
One composite surface water sample and one composite sediment sample (0-1 ft depth) were collected. The composites consisted of four grab samples, one from each corner of the pond. A downgradient monitoring well was installed to a depth of 15 feet in clayey silt and was screened over an interval of 5-15 feet.

22.2.2 Phase II Site Investigations

A ground water sample was collected from the monitoring well installed in Phase I. The purpose of this sampling was to determine if contaminants present in the pond have migrated to ground water. Due to an oversight, two piezometers which were scheduled for this site were not installed. One composite sample of pond sediment was collected for total and extractable chromium analysis. The purpose of this analysis was to

FIGURE 22-1

SITE 15-AREA 7 PLATING POND
SITE 16-AREA 7 INDUSTRIAL PARK
PHASE I



determine if the pond sediments represent a source of leachable chromium or exhibit the characteristic of a hazardous waste.

22.2.3 Site Hydrogeologic Characterization

22.2.3.1 Site Geology

Based on results of the test boring procedure, the subsurface unconsolidated overburden consists of a medium to dark brown clayey silt with some fine sand as was identified in Boring 15-4. This material is present from the ground surface to at least 15 ft. in depth (total depth of boring). Bedrock was not encountered in the boring and therefore depth to bedrock and bedrock lithology is not known. As only one monitoring well was installed, the lateral extent and variability of the overburden is also unknown.

22.2.3.2 Site Hydrogeology

Shallow ground water occurring beneath the site was found at a depth of 2 to 4.5 ft. below the ground surface within the clayey silt soil unit during June 1987. The monitoring well installed screened this upper water table. Ground water elevations collected during the winter and summer of 1987 (wet and dry seasons, respectively) indicate a water table fluctuation of 2.5 ft. with water levels dropping during the summer months (Table 4-3). Figure 34-4 illustrates the monitoring well location and the ground water elevation of 18 June, 1987.

22.3 Analytical Results (See Appendix I, Page 15)

22.3.1 Phase I Analytical Results:

The pond water analyses did not indicate the presence of contaminants at concentrations above Illinois General Use or Public Water

Supply Standards but Iron (1000 ug/L) exceeded the Federal MCL of 300 ug/L. The MCL for Iron was based on aesthetic concerns, however, and does not represent a risk to human health or the environment. The sediment contained 508 mg/kg chromium, but chromium was not detected in the water. The metals concentrations detected were estimated for screening purposes (Exhibit B). The phosphorus level was somewhat elevated (1621 mg/kg) in the sediment. The pesticide alpha-endosulfan was also detected in the sediment (811 ug/kg), but not in the pond water.

22.3.2 Phase II Analytical Results:

EP Toxicity chromium in the extract from the pond sediment was below the detection level of 0.05 mg/L, less than the 5.0 mg/L criteria for EP Toxicity (40 CFR 261.24). The sediment analysis for total chromium was not completed due to an oversight in laboratory scheduling. The ground water contained total and filtered chromium of 15 and less than 1 mg/L respectively. Filtered concentrations of arsenic, cadmium and lead were below detection limits in the ground water and thus were within the Federal and state standards. Total metals concentrations were also within the standards; total arsenic and lead were 7.6 and 22 ug/L, respectively, and total cadmium was below the detection level of 5 ug/L. No volatiles or pesticides were detected, but PCBs were detected at 0.01 ug/l which is above the AWQC for human health. All other concentrations in water were within the Federal and State drinking water standards.

22.4 Environmental Effects

22.4.1 Qualitative Assessment

This site was chosen for investigation based on its history of use for

receiving spent plating wastewater. Since there did not appear to be an outlet from the pond, the only viable transport mechanism for migration of contaminants off-site was ground water.

Phase I investigations detected traces of chromium, phosphorous and alpha-endosulfan in the sediment, with chromium and alpha-endosulfan exceeding the typical Refuge background levels. However, the pond water analysis did not indicate the presence of contaminants at concentrations above Illinois Public Water Supply Standards. Supported by Phase I results, the EP Toxicity test for chromium in the Phase II resampling of pond sediment, was below the detection level. The levels of chromium and cadmium in ground water were below Illinois Public Water Supply Standards, but unfiltered arsenic and lead levels were slightly higher than the standards. The filtered concentrations for arsenic and lead were below the detection limits. Trace PCB concentrations were detected in the ground water, but the results were questionable since PCBs were not detected in other samples from the site or from adjacent sites, and past land use activities in the area do not support their presence.

Human activities in this area are limited to occasional visits by authorized Refuge personnel and persons using the storage buildings in Area 7. The plating pond is located in an elevated area of the site, removed from the roadway, and restricted by dense vegetation and a barbed wire fence. Therefore, human exposures would be of a short-term, non-chronic nature. Under present conditions, it can be concluded that the site does not represent a significant risk of chemical exposure to either humans or wildlife; however, some remedial action may be warranted to permanently decommission the pond.

22.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

22.4.3 Analysis of Uncertainties

The information relied upon for evaluating Site 15 included a history of plating wastewater discharge to the pond, a site inspection and sample analyses.

Chemical residue information consisted of analytical results on one composite surface water sample and one composite sediment sample. Since contamination of the site occurred through the discharge of plating waste into a pond with no outlet, the area of contamination is likely to be limited to the water and the surrounding sediment.

It can be concluded that the data generated and the review of historical information are adequate for evaluation of the remedial alternatives for this site. The sampling analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

22.5 Preliminary Remedial Alternatives

The analytical results and risk evaluation presented in the previous sections indicated that this site does not contain contaminant levels that would be detrimental to the environment or to human or wildlife receptors. However, since the pond is no longer active, remedial measures for closure will be evaluated as part of the FS.

22.6 Conclusions and Recommendations

It can be concluded that although the plating pond site does not currently represent a significant risk of exposure at the Refuge, further evaluation is recommended as part of the FS to determine options for permanent closure of the pond.

SECTION 23 - SITE 16, AREA 7 INDUSTRIAL SITE

23.1 Site Description

Area 7 consists of a complex of 33 identical buildings over an area of 55 acres which have been used for a variety of industrial purposes during the past forty years. The buildings are arranged in six rows, each of which were at one time served by a railroad siding. Most of the buildings are currently used for dry warehousing purposes. However, three buildings are used by Pennzoil for waste oil recovery and recycling operations. Black residues are noticeable around some of these buildings. Two other buildings are used by a refurbisher of mining equipment. Black residues are also evident around these buildings. A drainage channel runs from south to north through the site.

Site 16 consists of the area in the vicinity of the five identified buildings and the drainage ditch (see Figures 22-1 and 23-1).

23.2 Site Investigations

23.2.1 Phase I Site Investigations:

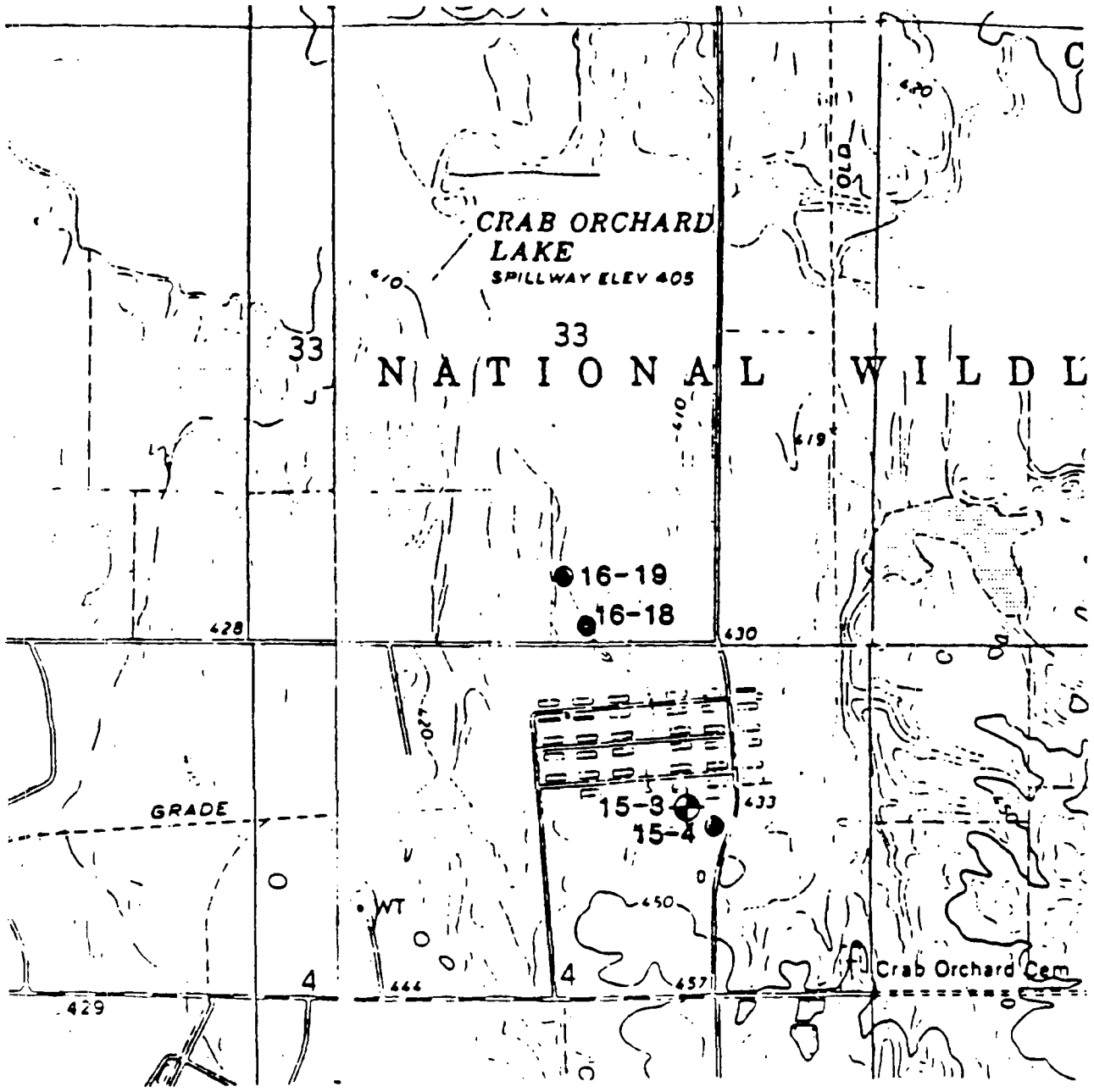
Two composite surface water samples and three composite sediment samples (0-1 ft depth) were collected from the drainage ditch. Nine composite soil samples (0-1 ft depth) were also collected from those areas which exhibited black oily soil residues in the vicinity of several manufacturing buildings. Three samples (one sediment and two soils) were resampled for full CLP organics analyses.

23.2.2 Phase II Site Investigations:

To define the areal exter of contaminant migration, one composite surface water and one composite sediment sample were collected from the

FIGURE 23-1

SITES 15 & 16
SAMPLING LOCATIONS
PHASE II



drainage ditch at locations downstream from the Phase I locations (See Figure 23-1). The Phase II samples were analyzed for magnesium, lead, arsenic, purgeables, pesticide/ PCBs, and base/neutrals/acid extractables. The purpose of these analyses was to determine if surface residuals from the Area 7 buildings were migrating along the ditch toward Crab Orchard Lake.

23.3 Analytical Results (See Appendix I, Page 16)

23.3.1 Phase I Analytical Results:

One of the two water samples from the drainage channel contained low concentrations of organics including chloroform (77 ug/L) and carbon tetrachloride (66 ug/L). The detected volatile compounds were above the AWQC for human health of 0.19 ug/L for chloroform and 0.4 ug/L for carbon tetrachloride but not above the correspondir criteria for aquatic life protection. Both samples contained magnesium (25 and 36 mg/L) and manganese (340 and 70 ug/L). No parameters were at concentrations above Illinois General Use Water Standards, but manganese levels were above the Illinois Public Water Supply Standards and the Federal Drinking Water MCLs. Manganese levels are not present at concentrations which would pose any concerns for public health. Two soils and one sediment selected for CLP organics contained trace organics (estimated concentrations below the detection limit or, in the case of acetone, detected in the QA/QC blank). The organics detected were, on a wet weight basis, acetone (348-782 ug/kg), N-nitrosodimethylamine (not detected to 115 ug/kg), crysene (not detected to 453 ug/kg), pyrene (not detected to 356 ug/kg), fluoranthene (not detected to 389 ug/kg), and low microgram levels of other base/neutral compounds,

possibly due to the oil spillage indicated by the black residues. Most of the organics detected were estimated values since the reported concentrations were below the analytical detection limits. In addition, the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present. Metals concentrations are estimated. All other concentrations in soil samples were consistent with those detected at the control sites.

23.3.2 Phase II Analytical Results:

Acetone and methylene chloride were detected in the water sample but at levels attributed to blank contamination. Aldrin (0.17 ug/L) and dieldrin (0.54 ug/L) were present at concentrations below the Illinois Public Water Supply Standards but above the AWQC for humans and for aquatic life. Holding times were exceeded for the volatiles analysis, and the recoveries for duplicate and spike samples were outside the QC limits for the volatiles and pesticide analyses. The dissolved magnesium concentration was 26,300 ug/L.

Traces of volatile compounds, including chloroform and chlorobenzene, were found in the sediment, as well as 1600 ug/kg di-n-butyl phthalate. The latter was also detected in the QA/QC method blank sample. No other organics were detected. Magnesium (958 mg/kg), lead (13 mg/kg) and arsenic (10 mg/kg) were detected in the sediment sample.

23.4 Environmental Effects

23.4.1 Qualitative Assessment

This site was chosen for investigation based on its past and current use as a site for storage activities, waste oil recovery, and recycling operations. Observable black residues, possibly originating from oil spills, suggested that this site might be a source of waste material. The drainage ditch provides a viable mechanism for transport of potential contaminants.

Phase I investigations detected chloroform and carbon tetrachloride in one water sample from the ditch. Both water samples contained magnesium but not at concentrations considered to threaten wildlife or affect human health. The soils and sediment samples from Phase I did not contain any contaminant concentrations that were significantly above Refuge background levels. Phase II data support the Phase I results; traces of aldrin and dieldrin were detected in the water sample but were below Illinois Public Water Supply Standards and Federal drinking water standards although above the more stringent AWQC for these parameters. Magnesium was again detected in the water but not at levels that would represent a concern. Magnesium, lead, and arsenic levels in sediments were similar to the concentrations found at the control sites. Di-n-butylphthalate was detected but was also present as a contaminant in the QA/QC blanks. N-nitrosodimethylamine was detected in Phase I but at levels similar to those at the munitions control site, or at approximately one half the levels determined to be acceptable for potentially exposed human or wildlife receptors at Site 17, Section 24.4.

The sampling results from both Phase I (upstream) and Phase II (downstream) did not indicate any evidence of migration of contaminants off-site. Since trace contaminants were detected, the presence of a waste

source may be justifiable; however, the small intermittent ditch does not support significant aquatic life, and human visitors to the site are very rare because the area is used for long-term storage and not for day-to-day industrial operations or residential purposes. Potential human exposures at the site might consist of occasional visits for 2-3 hours for purposes of loading or unloading supplies to storage. Since no receptors at the site could be exposed to chronic risk levels, and the levels of contaminants do not represent a risk for acute (short-term) exposure, it is not possible to have a "complete" exposure scenario. Therefore, on the basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

23.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

23.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a site inspection and the sample analyses. Black residues were noticeable around the buildings on the site, suggesting that they may be a source of contamination.

Chemical residue information consisted of analytical results on surface water, sediment and soil samples. This information was obtained only for the top one foot of soil; deeper soil borings and ground water monitoring were not conducted. Since any contamination of the site would

have occurred through a variety of spills and leaks during storage and loading rather than through excavation and burial, the area of contamination would be limited to the surface soil and sediment. The samples collected should adequately represent the conditions of the site.

It can be concluded that the data generated are adequate for evaluation of the remedial alternatives for this site. The sampling analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

23.5 Preliminary Remedial Alternatives

The analytical results discussed in the previous sections indicate that site constituents have not migrated and thus do not represent a risk for offsite transport of contaminants. The low levels of constituents found do not represent a threat of exposure to humans, aquatic life, or terrestrial wildlife. However, due to intermittent storage activities in this area, it is recommended that a follow-up monitoring program be conducted. Attachment 1 details the frequency and sampling plan recommended. The parameters of interest for monitoring in water include volatile organics (chloroform, carbon tetrachloride), semivolatiles, and pesticides (aldrin, dieldrin).

23.6 Conclusions and Recommendations

It can be concluded that the Area 7 Industrial Site does not pose a risk to humans or wildlife. No further evaluation in the FS is recommended, however, a monitoring program for volatiles, semivolatiles and pesticides in water (Attachment 1) might be implemented.

SECTION 24 - SITE 17, JOB CORPS LANDFILL

24.1 Site Description

Northeast of the Refuge Water Works is a ten-acre pond created by the Job Corps in the mid-1960s (See Figure 24-1). Attention was brought to this pond in early 1985 because thirty or more geese carcasses were found floating on the water or littering the shores. Some of these carcasses were relatively fresh while others were in various states of decay. The Fish and Wildlife Service has completed extensive analyses of these carcasses and has ruled out a variety of potential chemical causes although a definite conclusion was never reached. The "Job Corps Landfill" was discovered while investigating the geese kills.

Site 17 is comprised of the pond and adjacent one-acre landfill located north of the pond in a wooded area. The area contains widespread debris, such as bottles and cans, which do not appear to be buried deeply. Mica flakes, small electrical contacts, and a few small capacitors have been observed among the debris.

Aerial photos are available for the area near the Job Corps Site for 1951, 1960, 1965, 1971, and 1980. This sequence provides information on the usage of the Job Corps area. The 1951 photo shows a clearly defined fan-shaped dumping area with an access road entering from the southeast. This access road suggests that dumping in this area was conducted over an extended period. The access road is a well-defined one-lane path.

The 1960 photo indicates that the site was inactive at this time. The landfill area shows some vegetative cover and the contrast with surrounding vegetation is less distinct than in the earlier photos. The access road is still visible, but covered by brush in areas, suggesting that it had not been used

FIGURE 24-1

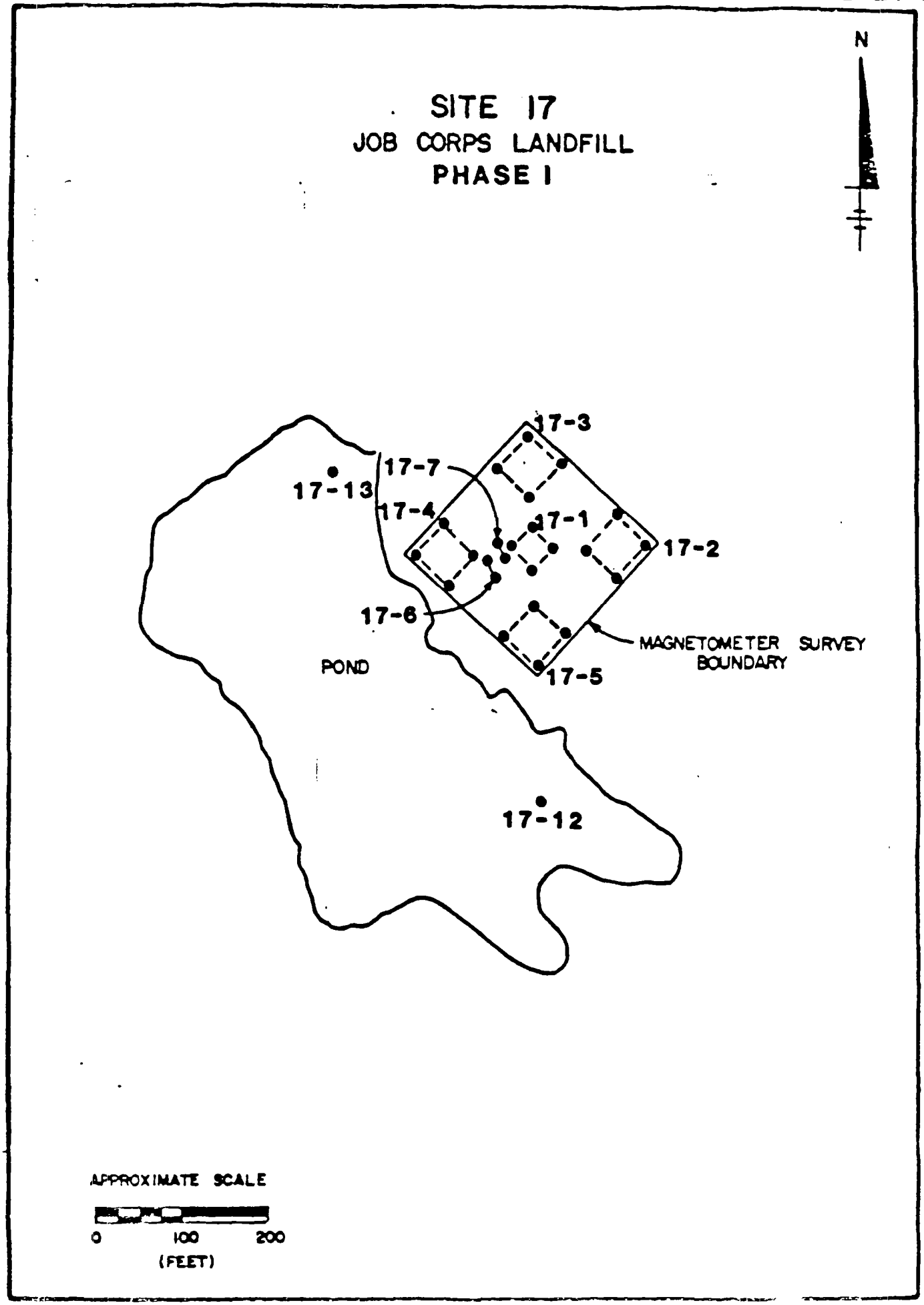
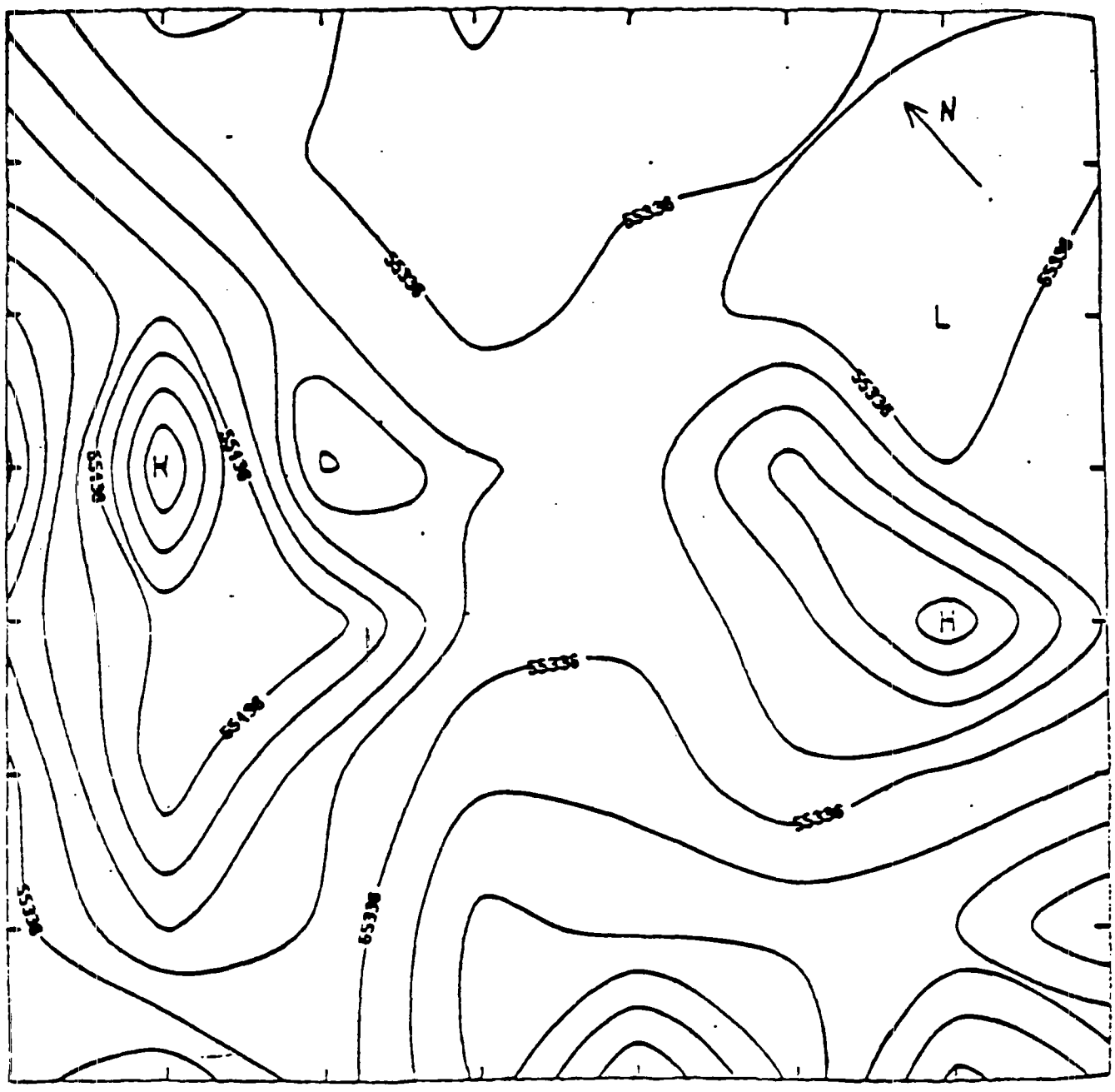

N
SITE 17
JOB CORPS LANDFILL
PHASE I
17-3
17-7
17-13
17-4
17-1
17-2
17-6
MAGNETOMETER SURVEY BOUNDARY
POND
17-5
17-12
APPROXIMATE SCALE
0
100
200
(FEET)

FIGURE 24-2

SITE 17

MAGNETOMETER SURVEY



for some time. The 1951-1960 timeframe is consistent with a 1957 automobile license plate observed during the site inspection.

In the 1965 photograph the access road is distinguishable only as a boundary of an adjacent planted field. The landfill area is not covered by brush or trees, and the vegetative cover appears similar to that in the surrounding area.

The 1971 photo shows the Job Corps pond which was created in the mid-1960s. The eastern end of the pond overlies an area which had been the access road. The landfill area shows signs of trees, brush and larger vegetation. The 1980 photo shows the site much as it is today, with trees and heavy vegetation overlying the fill area.

During the site inspection, various articles typically associated with municipal refuse were noted, including bottles, plates, etc. The site is presently covered with dense vegetation and the access road is no longer useable.

24.2 Site Investigations

24.2.1 Phase I Site Investigations:

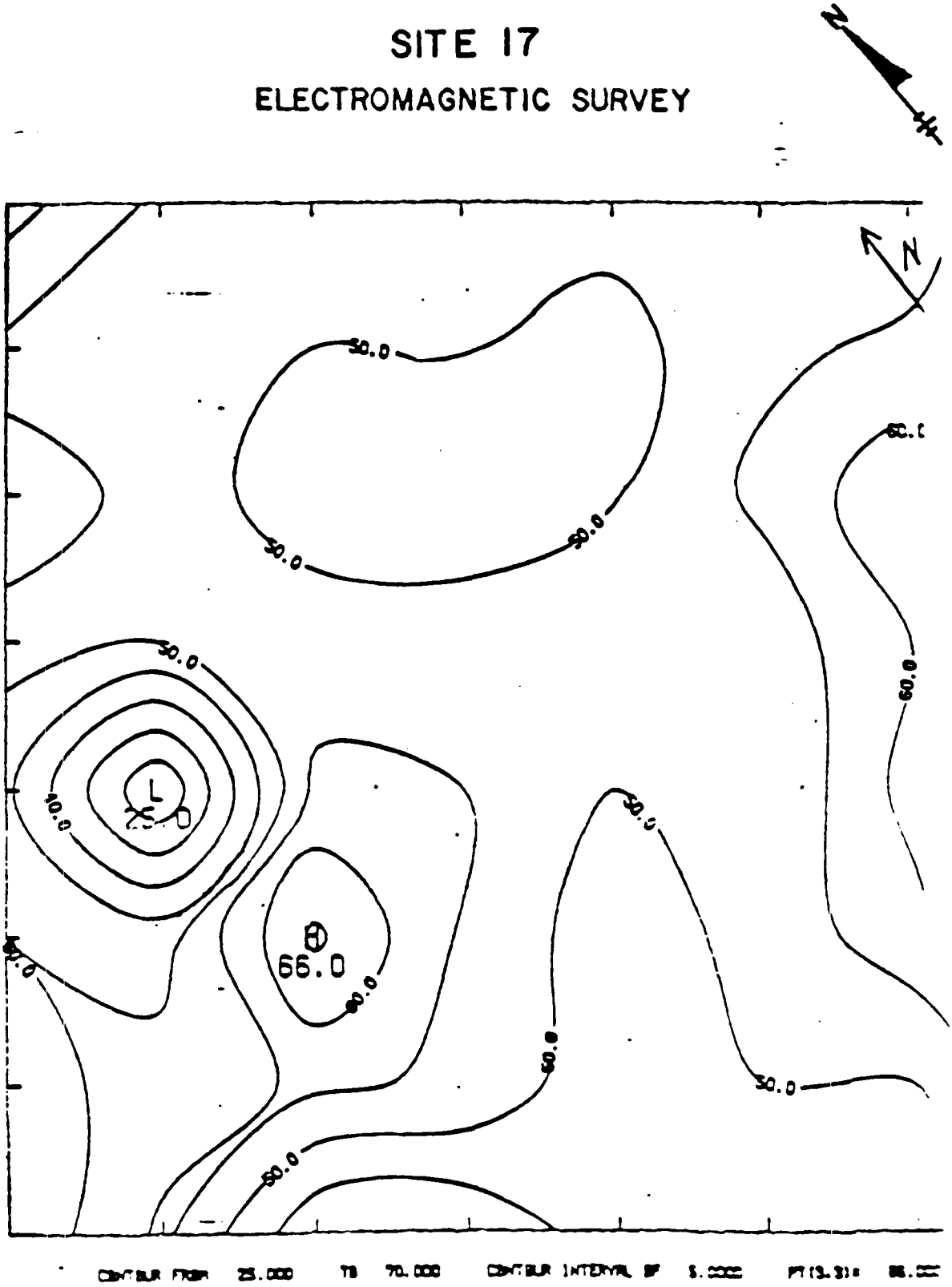
A magnetometer and electromagnetic terrain conductivity survey was conducted. The results of these surveys are shown on Figures 24-2 and 24-3.

Two composite surface water samples were collected from the pond, and seven composite soil samples (0-1 ft depth) were collected from the landfill. A composite was collected from each of five 50 ft x 50 ft squares within the landfill as well as from two bare patches. Figure 24-1 depicts the Phase I sampling locations.

FIGURE 24-3

SITE 17

ELECTROMAGNETIC SURVEY



24.2.2 Phase II Site Investigations:

Phase II studies were intended to better define the vertical and horizontal extent of the contamination identified in Phase I (See Figure 24-4). Forty-seven soil samples were collected at the landfill, thirty-five at depths of 0-1 ft and twelve to a depth of 3 ft. All soils were analyzed for PCBs, lead and cadmium. In addition, twelve of these samples were also analyzed for explosives and nitrosamines. One Phase I soil (location 17-1) was reanalyzed for mercury. Two composite surface water and six grab sediment samples were collected from the pond and analyzed for PCBs, lead, cadmium and explosives. The pond water samples were also analyzed for CLP organics, PCBs, arsenic, and copper. Three surface (0-1 ft. depth) sediment samples were collected along the shallow embankment and three were collected from the deeper area of the pond.

Five monitoring wells, four shallow and one deep, were installed to depths of 12 to 30 feet (see Figure 24-4). The wells were set in silt and silty clay soils, except well 17-65 which was set on top of rock. Each well utilized five-foot stainless steel screens. Ground water samples were analyzed for CLP HSL organics and metals, low-level PCBs, nitrosamines and explosives.

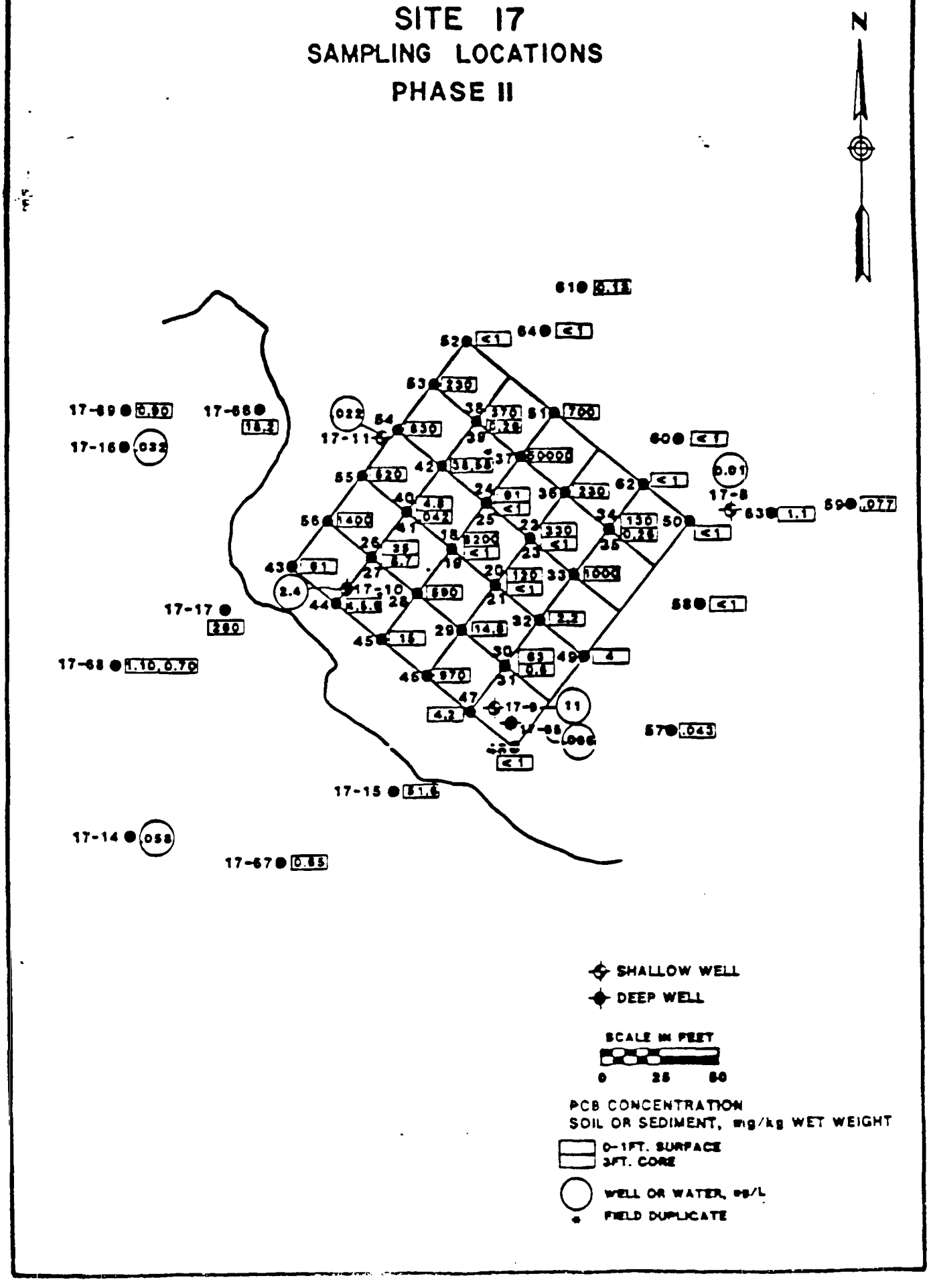
Two composite fish (one bass and one bluegill) were sampled from the Job Corps Pond shortly after the Phase II investigations. The samples were analyzed for pesticides, PCBs, cadmium, mercury and lead.

24.2.3 Site Hydrogeologic Characterization

24.2.3.1 Site Geology

The sequence of unconsolidated soils underlying the site to the maximum penetrated depth of 28 feet (Boring 17-65) consist of predominantly silty clay. Occasional organic silt layers with a trace of

FIGURE 24-4


SITE 17
SAMPLING LOCATIONS
PHASE II
N
SHALLOW WELL
DEEP WELL
SCALE IN FEET
0
25
50
PCB CONCENTRATION
SOIL OR SEDIMENT, mg/kg WET WEIGHT
0-1FT. SURFACE
3FT. CORE
WELL OR WATER, ug/L
FIELD DUPLICATE

fine gravel were identified in upper portions of Borings 17-9 and 17-10 and near a small swamp south of the site.

Underlying the silt and clay sequence in Boring 17-65, a shale bedrock was encountered that was gray and friable. The depth to bedrock was not identified in other borings.

24.2.3.2 Site Hydrogeology

Occurrence of Ground Water

Ground water was identified to occur about 6 to 8 feet below ground surface during June 1987 in the shallow wells. Ground water levels were about 5 feet higher during December 1986. The ground water occurrence in the deeper well set on top of bedrock exhibited a slightly lower hydraulic head.

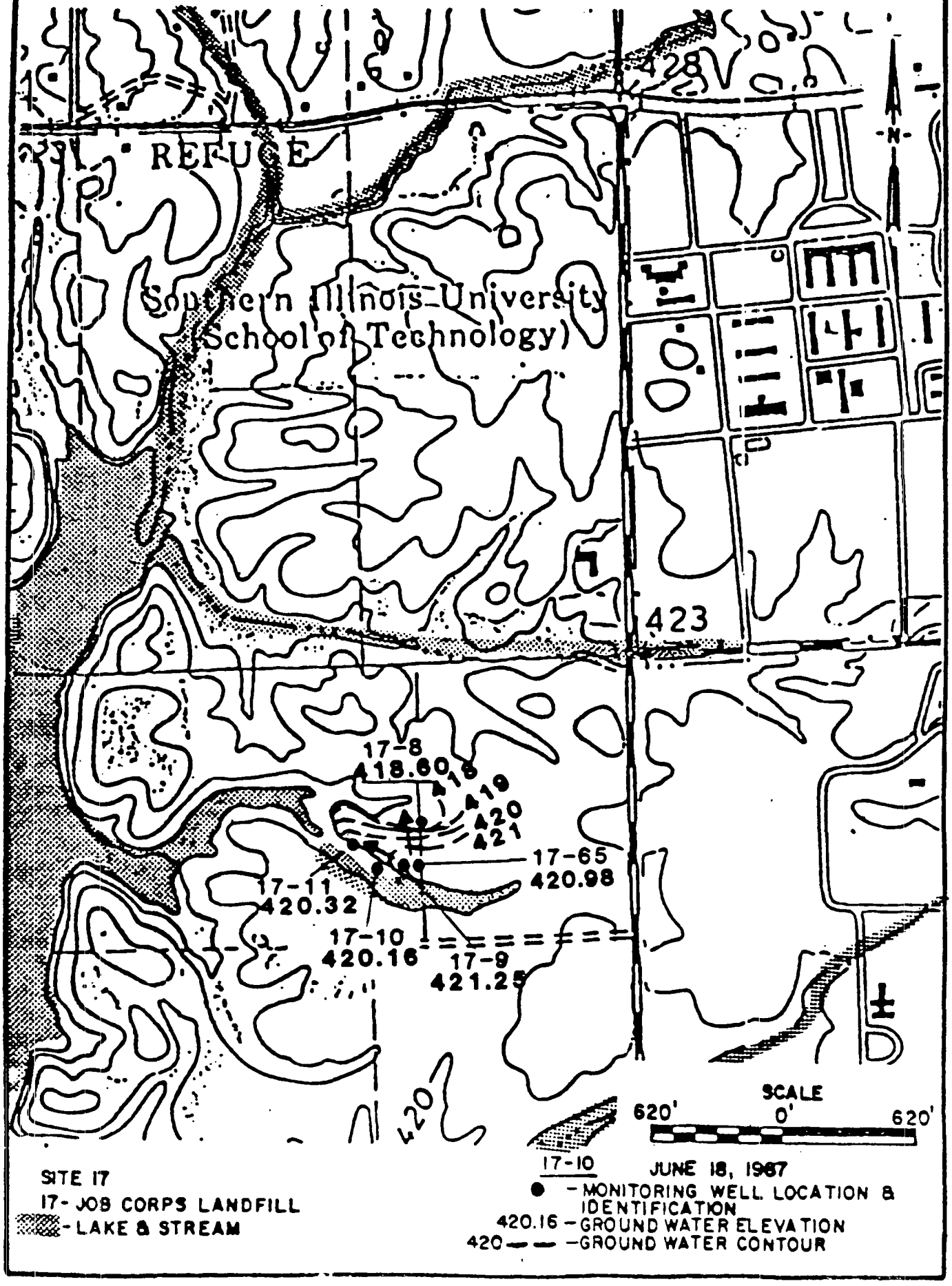
Ground Water Flow Conditions

Elevations of ground water within the water table wells were contoured and are shown on Figure 24-5. Results of these data taken from June 1987 indicate a radial groundwater flow pattern to occur in north, west, and south directions toward adjacent surface water drainage swales. Discharge from the swales which contain intermittent surface flows is toward Crab Orchard Lake about 1,000 feet west.

Ground water velocity calculations employing the formula given in Section 4.2 were performed utilizing an average hydraulic gradient of 0.2 ft/ft (June 1987), average hydraulic conductivity (K) of 9.36 ft/day, and a porosity of 0.35 (Davis, 1966). Resultant ground water flow velocity was calculated to be about 9.36 ft/day or 3,400 ft/year.

FIGURE 24-5

SITE 17 GROUNDWATER FLOW MAP



Vertical flow potential from the upper water bearing zone was found to be slightly downward into lower unconsolidated aquifers based on ground water elevations taken from well nest 17-65 17-9.

24.3 Analytical Results (See Appendix I, Page 17)

24.3.1 Phase I Analytical Results:

The magnetometer and electromagnetic survey shown in Figures 24-2 and 24-3 of the landfill did not reveal the presence of any concentrated pockets of conductive materials.

The surface soil samples collected from depths of 0-1 ft. from the landfill showed the widespread presence of PCBs and lead. All surface soil samples contained PCBs (Aroclor 1254), with concentrations ranging from 21 to 1700 mg/kg wet weight (28 to 2059 mg/kg dry weight). Lead was also present in all surface soil samples at concentrations of 609 to 14100 mg/kg wet weight. The detected PCB and lead levels at this site are not typical of the Refuge background. One soil sample contained 34 mg/kg cadmium and one soil contained 3 ug/kg mercury. All other metals were within the range of concentrations detected at the control sites. The metals concentrations performed in Phase I are estimated for screening purposes.

The explosive tetryl was detected in four soil samples at concentrations ranging from 3.7 to 6.5 mg/kg. One of the soil samples (location 17-2) analyzed for CLP organics contained on a wet weight basis, di-n-octyl phthalate (4760 ug/kg), 1984 ug/kg methylene chloride, and 173 ug/kg acetone. The FID scan of the same samples showed 127,895 ug/kg. Methylene chloride and acetone parameters were also present in the method blank. The second soil sample (17-6) analyzed for CLP organics (with 654,308 ug/kg FID scan) contained 220 ug/kg wet weight

N-nitrosodimethylamine, but all other organics detected were below the detection limit. One water sample contained 2.6 ug/L of 2, 4-DNT explosive residue, which is above the human health AWQC of 0.11 ug/L but below the aquatic life 24-hr average criterion of 230 ug/L. No explosives were detected in the other water sample or in the sediments or soils. PCBs and all other organics were undetected (below 10 ug/L) in the pond water samples, and all analyzed compounds were within the Illinois Public Water Supply Standards. It should be noted that the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data. (See Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present.

24.3.2 Phase II Analytical Results

PCB concentrations in surface (0-1 ft. depth) soils ranged from 0.077 to 50,000 mg/kg wet weight (0.101 to 69,042 mg/kg dry weight). Soil cores collected at 3 ft. depths, however, showed PCB concentrations below 1 mg/kg wet weight for all except one core (17-27) which contained 8.7 mg/kg wet weight (11.6 mg/kg dry weight). Figure 24-4 shows the areal distribution of PCB concentrations throughout the site on a wet weight basis. Lead concentration in soils ranged from below 6 to 17414 mg/kg (dry weight) in the 0-1 ft. samples, and from below 6 to 13 mg/kg (dry weight) at 3 ft. depth with the exception of soil 17-27 which had 219 mg/kg lead. The highest lead concentrations were found to coincide with the highest PCB levels. Cadmium concentrations were relatively low and heterogenous. At the 0-1 ft strata, eight soil samples showed less than 1 mg/kg cadmium, while nineteen samples showed cadmium from 1 to

57 mg/kg, with a mean of 16 mg/kg. Cadmium concentrations were less than the detection limit of 1 mg/kg at the 3 ft. depth soil samples. The soil sample from Phase I (location 17-1) that was reanalyzed for mercury contained 190 ug/kg. No explosive residues were found in the soils. The pond bottom surface sediments contained between 0.65 and 260 mg/kg wet weight (3 and 607 mg/kg dry weight) PCBs, 4-49 mg/kg cadmium, and 38 to 5722 mg/kg lead. No other parameters were detected.

The groundwater and pond water samples contained traces of di-n-butyl-phthalate (2-12 ug/L, below the AWQC for human health of 35,000 ug/L), isophorone (40-115 ug/L, below the AWQC of 5,200-117,000 ug/L), acetone (22-4,270 ug/L), and methylene chloride (2-56 ug/L). Acetone, isophorone, and methylene chloride were also detected in the method blanks. The organics analyses were outside the QC limits for duplicate recoveries for some compounds were extracted outside of holding time.

The explosive nitrobenzene was detected in the water samples but not in the soils or pond sediments. Nitrobenzene concentrations (unfiltered) were 1.48 to 11.4 ug/L in the wells and 1.08-1.68 ug/L in the pond water, well below the AWQC of 19,800 ug/L (human health level), or 27,000 ug/L (aquatic life acute maximum).

Cadmium was below the 5 ug/L detection level (total and dissolved) for all pond and groundwater samples. Arsenic ranged from 7 to 27 ug/L total, and from less than 2 to the detection limit of 25 ug/L dissolved. Chromium ranged from 18 to 139 ug/L total and 1.7 to 6.4 ug/L dissolved in the wells, exceeding the standard of 50 ug/L for Illinois public waters and Federal drinking waters; chromium was not analyzed in the pond

water samples. Lead was detected between 4.3 to 55 ug/L total and 3.8 to 21 ug/L dissolved in groundwater, and between 1.6-3.6 ug/L total and less than 1 ug/L dissolved in the pond. Only one sample, ground water from well 17-11, exceeded the Illinois and Federal standard of 50 ug/L for lead. The ratios found between total and dissolved metals values indicated that the higher concentrations detected are mostly associated with suspended solids. All dissolved metals concentrations were below the Illinois Public Water Supply standards and Federal MCL standards.

PCBs (total aroclors) were detected in the groundwater samples at 0.01 to 11 ug/L. The higher range detected may have been due to suspended matter in the sample since most of the detected concentrations are higher than the solubility limit of PCBs in water. The pond waters contained 0.032 and 0.058 ug/L PCBs. All detected PCB concentrations were above the AWQC for human health and aquatic life protection. One well (17-65) contained traces of chloroform (12.5 ug/L), another well (17-10) had pentachlorophenol (19 ug/L), and bis(2-ethylhexyl) phthalate (23 ug/L). Well 17-9 also contained 10 ug/L of bis(2-ethylhexyl) phthalate. Chloroform levels exceeded the AWQC for human health of 0.19 ug/L; pentachlorophenol was above the criteria for protection of aquatic life (3.2 ug/L average), but not above the level for protection of human health (1,010 ug/L). The concentrations of phthalates were below the applicable AWQC criteria of 15,000 ug/L (human health). Two fish (a bass and a bluegill) sampled from Job Corps Pond were analyzed for pesticides, PCBs, and metals. No pesticides or PCBs were detected, but low levels of cadmium (0.05 mg/kg in bluegill only), lead (0.57 mg/kg in bass, 6.94 mg/kg in bluegill), and mercury (0.243 mg/kg in bass, 0.072 mg/kg in bluegill) were found.

Two composite fish (one bass and one bluegill) sampled from the Job Corps Pond were analyzed for pesticides, PCBs, cadmium, mercury and lead. No pesticides or PCBs were detected (detection limits 20 to 400 ug/kg). Low levels of cadmium (0.05 mg/kg in bluegill only) and mercury (0.243 mg/kg in bass, 0.072 mg/kg in bluegill) were detected. Lead concentrations were higher for bluegill, at 6.94 mg/kg, while only 0.57 mg/kg were detected in bass.

24.4 Environmental Effects - Job Corps Landfill

24.4.1 Qualitative Assessment

24.4.1.1 Source Evaluation

As described in Section 24.1, the site was likely to have been an area where mixed municipal and industrial type wastes were disposed. Field inspections of the site revealed various cans, bottles, containers and other articles typical of a mixed sanitary waste. However, the site inspection also noted the existence of a small exposed area containing wire electrodes and electrical components, suggesting that this location may have also received wastes of industrial origin. Geophysical investigations suggested widely-scattered debris rather than concentrated areas of fill material. The soil boring program encountered materials just below the surface (0-1 ft) consistent with what was observed at the surface.

Results of the soil and sediment sampling and analysis program indicate that the primary chemical compounds of concern at this location are PCBs, cadmium, and lead. Levels of cadmium at the site are very heterogenous making it difficult to derive accurate value for exposure in a quantitative risk assessment. In view of concern for cadmium toxicity to wildlife a qualitative risk assessment was completed. The

concentrations of PCBs detected, ranging from 0.077 to 50,000 mg/kg wet weight (0.10 to 69,042 mg/kg dry weight) are unusually high and inconsistent with what might be expected to be present within a landfill containing municipal wastes. Of the chemical compounds included and consistently detected at elevated levels in the sampling and analysis program, PCBs and lead are the most toxic. Therefore, for purposes of the quantitative risk assessment, it will be assumed that PCBs and lead are the compounds representing the highest risk. Since N-nitrosodimethylamine was also detected in a soil sample collected during the Phase I investigation, it will also be considered in the quantitative risk assessment for this site. Furthermore, for purposes of the nitrosamines assessment, and in order to provide a worst case conservative outcome in light of the uncertainty associated with the analytical result for this specific class of compounds, the concentration of nitrosamines in soil will be considered to be twice the level found in the single soil sample analyzed (i.e. 440 ug/kg rather than the detected level of 220 ug/kg). PCBs, lead and N-nitrosodimethylamine compounds are therefore used to define the waste "source". Exposure to cadmium in soil and sediment will qualitatively follow the exposure scenarios of PCBs and lead.

The physiochemical and toxicological properties of PCBs, nitrosamines and lead, as well as cadmium are summarized in Exhibit A. Lifetime dietary exposures of rodents to PCBs have established the carcinogenic potential of these compounds. PCBs produce a number of other chronic and subchronic effects as well. Lead exposure also presents cause for concern due to neurological, hematological and other effects demonstrated in humans and animals produced by chronic exposure to low lead levels.

There is insufficient evidence to determine if lead exposure presents a risk of cancer. Although dietary lead acetate has produced cancer in test animals, lead acetate would not be expected at the site due to chemical conversion to insoluble oxide and sulfate salts (USEPA ECAO-CIN-H055, 1984). PCBs were detected in surface soils in the landfill, as well as in the sediments of the adjacent pond. Based on the physiochemical properties of PCBs and lead, it can be assumed that they are strongly adsorbed to the surfaces of soil materials. They will therefore not be leached by runoff or surface infiltration, but will behave and be transported along with the particles to which they are adsorbed. There were no free liquid PCBs encountered during the investigation.

Because of the identification of PCBs in all of the soil samples collected, it is assumed that all of the samples within the defined boundaries of the site contain PCBs at levels in the range detected. For purposes of the quantitative risk assessment, the average value of 7,950 mg/kg dry weight PCB as Aroclor 1254 will be used as the worst case mean concentration, and the maximum concentration detected (69,042 mg/kg dry weight Aroclor 1254) will be used to represent the "upper bound" worst case residue. For lead, a soil level of 5,000 mg/kg dry weight was selected as a representative upper bound value for the site.

24.4.1.2 Transport Route Evaluation

a) Air: PCBs in soil exert a relatively low vapor pressure, but under certain conditions, transport of PCBs in the vapor state could constitute a functional transport route. Lead does not exert an appreciable vapor pressure, and therefore cannot be transported in the vapor state in significant quantities. However, because of the

existence of exposed, soil-adsorbed PCBs and lead at the site, dusts generated by wind erosion, vehicular traffic or activities by endogenous wildlife are likely to contain contaminants which can subsequently be transported by the air route. Therefore, it can be concluded that the air transport route can function to carry PCBs and lead and other waste related compounds to on- and off-site locations for subsequent exposures by receptors in those areas.

b) Direct Contact: Because of the existence of PCBs and lead in exposed wastes, soils, vegetation and sediments in the area, exposures by the direct contact route are possible.

c) Surface Water: There is a pond located directly adjacent to the waste area. The pond discharges to an adjacent stream through an engineered control weir, also adjacent to the waste area. Inspection of the site revealed the presence of waste materials at the shoreline, as well as submerged in the pond. The surface of the wastes in the site are approximately zero to four feet above the level of the pond. Therefore, it is likely that runoff from the waste site is directed into the adjacent pond.

In addition to the proximity of the exposed waste materials to surface water, the results of the sampling and analysis program indicate that the pond sediments contain PCBs. It is presumed that the PCBs in the pond sediments originated from the materials deposited in the landfill. On this basis,it can be concluded that the surface water transport route can function to transport waste materials from within the site to the pond, as well as by stream flow to offsite locations.

d) Ground Water: Results of analyses of the ground water indicate only minor residues of PCBs ranging from 0.01 ug/L to 11 ug/L. Lead was found at 4.3 mg/L to 55 mg/L. However, it is likely that the contaminants detected in the ground water samples are the result of the installation of the monitoring wells within a fill area containing PCBs and lead, rather than PCBs and lead which have migrated with the movement of water through the active mass. This behavior is consistent with what is known and previously observed regarding the movement of PCBs and lead in ground water and the absence of these constituents in subsurface soil. They are not expected to move with ground water to any appreciable extent due to their high affinity towards silty soils observed at this site. (See hydrogeology discussion, Section 24.2.3). The measured soil hydraulic conductivity at this site ranges from 1.5×10^{-6} to 2.64×10^{-4} ft/sec, (Table 4-4) further indicating the retardation of movement of PCBs and lead.

Based on the detection of only minor contaminant residues in the ground water, and, more importantly, on what is known regarding their behavior in ground water, the ground water transport route is not considered a significant means of transporting PCBs and lead to offsite locations. Because residues of the other monitored components such as cadmium were not detected in the ground water, and there are no ground water users in this area, the ground water transport route will not be considered further as a component of the risk assessment.

24.4.1.3 Receptor Evaluation

Human

As described in the section on the general land use within and adjacent to the Refuge, the area within the Refuge is not populated by humans. There are only minor numbers of humans occupying the Refuge for industrial (occupational) purposes, and the Refuge has only a moderate daytime recreational use load. A large proportion of the refuge is designated as a wildlife sanctuary, and is therefore posted off limits to human use.

The area in the direct vicinity of the Job Corps site is not used as a manufacturing area. There are no users of ground water in this area. Access to the landfill has been minimized by cultivating tall and thick brush and trees in the area. Therefore, exposures by humans on a day to day basis would not be expected.

However, because of the existence of open water at this location, and its attractiveness to recreational users of the Refuge, a small number of human receptors could potentially experience potential one-time or isolated multiple exposure. This area of the Refuge is open for deer hunting for only one week per year; it is closed to the public for the remainder of the year. On this basis a human receptor could be characterized as a hiker or hunter. Also included as a human receptor would be a Refuge worker who might visit the area as part of a routine maintenance of the spillway.

It is likely that the total number of human receptors is low. A upper bound estimate of the number of human use-days in the pond area might be on the order of 25 per year according to the Refuge Manager. One human use-day is defined as a period of about four hours, where the pond and adjacent areas are used by

one human for recreational purposes. However, because the adjacent areas are more attractive to recreational users, the actual landfill area would experience a much lower amount of use days. An upperbound estimate of human use in the landfill, as related to hiking through the landfill on the way to a more "attractive" recreational area might be 10 use-days per year. Specific scenarios under which the human users of the area might become exposed to the PCBs, cadmium or lead residues in the area will be developed in the following sections.

The transport route evaluation identified three major functional transport mechanisms: the air route, the direct contact route, and the surface water transport route. Human recreational receptors identified in the previous section would be within the influence of each of these transport routes. However, for the most part, they would only be able to experience exposures while within the area of the landfill and adjacent pond, or as a consequence of their use of that area. There is no route for ingestion of contaminated fish unless a human receptor was to illegally catch and consume fish from the pond. Humans at areas removed from the landfill and pond, such as residents of nearby communities or other populated areas, as well as recreational users of the Refuge not entering the landfill or pond, would not be within the influence of the site.

The following sections describe the most likely exposure scenarios associated with each of the identified functional transport routes.

a) Direct Contact: The most likely human exposure scenario would be exposures to PCBs, lead, cadmium, and other waste components as a

result of direct contact with the waste materials and contaminated pond sediments. Because there are no human populations residing in the areas of the landfill, chronic (repeated long term) exposures would likely not take place. Direct contact exposures would occur on a one-time, or limited multiple event short term basis, experienced by recreational users of the pond and adjacent landfill.

Although PCBs and lead are not especially permeable through intact human skin, the most likely route by which they could enter the receptor's body following exposure would be through incidental ingestion of the soils or vegetation on the receptors body. Direct contact exposures could also extend into the users' homes, providing exposures to secondary receptors such as family and friends. It is possible that waste residues might become adhered to the shoes or clothing of a recreational user, to be transported to the user's home and become established as residues. The introduction of contaminated materials into a home along with soiled clothing or shoes could establish a reservoir of material that would persist and represent a longer term, chronic exposure.

b) Air Route: Dusts generated by wind erosion or foot traffic, as well as volatilized residues, represent a source and pathway for exposures via the air route. The most likely human receptors experiencing exposures via this route would include hikers or recreational users who would disturb and breathe contaminated dusts kicked up from the ground or dislodged from broad-leaf vegetation while walking through the landfill or adjacent areas. These exposures are likely to occur on an acute (one-time) basis or on a few occasions. Due to the absence of human populations near the

site, no chronic exposures would be expected. However, long distance transport of dusts towards populated areas is unlikely due to the level of vegetation at the site. Also possible, would be the inhalation of dusts by the recreational users and secondary receptors such as family members, who breathe dusts arising from soils adhered to the boots and clothing of the recreational user and transported out of the landfill.

c) Surface Water Route: The major uptake route usually associated with surface water exposures, such as by swimming and wading, is the inadvertent ingestion of water and sediment. However, the surface water route would not be a major human exposure route at this location because the pond is not used by humans for swimming or wading. There are other areas of the Refuge, notably Crab Orchard Lake, which are more attractive for swimming and wading. Nevertheless, it is possible that contact with the pond water and sediments might occur on a less than whole body basis, as might be associated with fishing and hiking. Neither fishing nor hiking are practiced in the immediate area. This route will be further evaluated within the quantitative assessment.

d) Ingestion: Ingestion of fish and game in the Job Corps landfill or the adjacent pond by human recreational users is an exposure pathway which requires further consideration. PCBs are lipophilic compounds which tend to partition into and accumulate in fat-containing tissues of animals. Ingestion of fish and game by humans utilizing the area near the Job Corps landfill would therefore be a potential scenario of exposure and uptake of PCBs as well as lead. However, only lead, and traces of cadmium and mercury were

found in a limited sampling of pond fish; PCBs were not detected in this sampling. A 1982 survey of contaminants in deer tissue at the Refuge (Ruelle, March 1983) did not show measurable PCB levels in either the fat or red meat tissues analyzed. A 1980 survey (Gritman, 1982) detected average lead levels in deer liver tissue at 5.6 mg/kg.

A relatively large percentage of the human users of the Refuge would be within a mile or so of the area for purposes of hunting. Fishing in the Job Corps site pond is not permitted although illegal fishing is conceivable. The abundance of game fish in Crab Orchard Lake and other lakes nearby further discourages fishing at the Job Corps pond. Because game would be taken from the area only on isolated occasions, this route would not represent a chronic exposure, but would be limited to a single acute or multiple acute exposures. Due to very the low likelihood of repeated use of the pond for fishing, human exposures by the ingestion of fish will not be considered further.

Wildlife

As described in the introduction to this report, Crab Orchard National Wildlife Refuge is an area that has been set aside to foster the breeding and preservation of wildlife endemic to that part of the country. Based on field inspections of the Refuge, as well as accounts relayed by Refuge Managers, there is an abundance of wildlife in the area. Of special note are the large populations of migratory aquatic waterfowl, including various species of ducks and geese who use the Refuge as a stopover during their excursions

north. Also of note are populations of white-tailed deer and numerous species of small mammals such as raccoon, rabbit and opossum. Crab Orchard Lake is also abundant with various species of warm-water fish, such as bass and catfish.

The area in the direct vicinity of the Job Corps Landfill is similar to the other areas of the Refuge. Of special note is the pond which is directly adjacent to the landfill. Field inspection of this location noted that the pond contained species of panfish, turtles, frogs, aquatic insects, and abundant aquatic vegetation. The pond was observed to be frequented by transient waterfowl. During the inspection, roughly 30 geese carcasses were found in the area. There were also deer tracks and a number of small mammal burrows.

With the exception of the general field inspections, there has been no formal survey of wildlife conducted on the area of the landfill and the adjacent pond. According to Ruelle (1987), the Refuge contains two active nesting areas of the endangered bald eagle. It could not be determined whether, in addition to common species of endemic wildlife, there were any other threatened or endangered wildlife species in the area. However, because many of the areas of the Refuge have been set aside as wildlife breeding grounds and also as stopovers for migratory birds, it is possible that other threatened or endangered species may be present in the Refuge.

Based on the field observations, as well as the high wildlife density in other portions of the Refuge, it is concluded that the Job Corps landfill and the adjacent pond contain wildlife receptors which

warrant consideration within this risk assessment. This area contains open water attractive to water-fowl, and the pond is of adequate area and depth to support aquatic life. The landfill area has become overgrown with vines which may provide a desirable location for the establishment of dens by small mammals. Therefore, it can be concluded that the landill and adjacent pond can, and most likely do, provide a transient as well as permanent habitat for various species of wildlife.

The following sections present exposure scenarios by each of the transport routes identified in the preceding sections as functional.

a) Air Route: Wildlife in the area would experience exposures to waste components by the air route. However, in contrast to the activities described for the human exposure, wildlife would be expected to come into more intimate contact with waste contaminated dusts and vapors through burrowing, preening and feeding activities. Therefore, inhalation exposures would be likely to be of considerable duration and frequency. In contrast to the limited frequency of exposures experienced by the human receptors, wildlife would experience both acute (in the case of transient species) as well as lifetime chronic exposures (in the case of endemic species). The magnitude and significance of these exposures will be discussed in the quantitative risk assessment.

b) Direct Contact: As with the inhalation route, wildlife in the area would experience chronic as well as acute exposures to the PCBs, cadmium, and lead in the landfill and the adjacent pond. These exposures would result in prolonged skin contact with the soil-borne

wastes, as well as ingestion of the materials through preening activities.

c) Ingestion: In contrast to the acute exposures potentially experienced by human users of the area, wildlife inhabiting the site would be likely to experience both acute as well as chronic exposures to PCBs, cadmium, and lead and other waste components via ingestion. These exposures would result from incidental ingestion of soils, vegetation, or sediments during preening and dusting (birds and small rodents) activities, as discussed in the direct contact section. Ground-feeding birds and gallinaceous birds (e.g. quail) ingesting grit could also ingest quantities of contaminated soil. Exposure would also occur as a result of normal dietary ingestion of vegetation and wildlife in the area.

Soils and dusts containing waste components would be consumed by herbivores along with contaminants accumulated in terrestrial or aquatic vegetation. Many aquatic waterfowl also consume large quantities of sediment as part of their diet. These inputs could represent a major exposure and uptake route.

With regards to animal life, aquatic and terrestrial inhabitants would be expected to contain bioaccumulated residues of PCBs, cadmium and lead. These bioaccumulated residues would be consumed by predatory wildlife, resulting in ingestion exposures. A limited survey of Job Corps Pond fish residues did not, however, detect PCB residues.

24.4.2 Quantitative Assessment

24.4.2.1 Estimates of Release and Exposure Rates

Estimates of Airborne Exposures

The qualitative section of this assessment has established that the air pathway represents a complete exposure route. This pathway consists of the breathing of PCB-contaminated dusts at the Job Corps landfill site created by occasional human recreational activities (i.e., hiking, hunting) and by the burrowing activities of wildlife, as well as PCB vapors evaporating from the soil. No air monitoring surveys were conducted, (except for the purpose of worker safety) nor were appropriate models available to estimate the quantity of respirable dust kicked up or dislodged into the breathing zone of a walking human.

Hwang et. al. (1986) developed one methodological approach for modeling exposure to airborne PCB residues which have evaporated from the adsorbed or liquid state on or in soils. Using the vapor pressure as a measure of the air concentration of Aroclor 1254 and the estimated air-soil partitioning behavior of PCBs, these authors reasoned that, for example, soil concentrations of Aroclor 1254 in excess of approximately 4 mg/kg would be a saturation concentration. Further, under such conditions the volatilization of PCBs can be estimated purely by consideration of the vapor pressure, without consideration of the soil adsorptive properties of the residues. In addition, the mass flux would thus be independent of the PCB concentration in the soil.

Hwang et al. (1986) estimate that the emission rate for Aroclor 1254 under saturated soil conditions is 1.13×10^{-10} g/sec/cm^2, or 1.13 ug/sec/m^2. Since the soil PCB concentration at this site exceeds the 4 ug/g concentration stated to represent a saturated condition in soil, the

emission rate given above will be used to model inhalation of PCB vapors on the site. U.S. EPA (1986) presents a model for estimation of airborne residues emitted from a hazardous waste site which considers both advection and dispersion. Conceptually, the model is comparable to the methodology employed by Hwang et al. to perform a similar estimate. However, because U.S. EPA (1986) presents the methodology in greater detail, it will be utilized here. The basic relationship is:

$$C(x) = \frac{Q}{pi*y*z*u}$$

where:

C(x) = is the concentration of substance at distance x from site (mass/volume)

Q = release rate of substance from site (mass/time)

y = dispersion coefficient in the lateral (crosswind) direction (distance)

z = dispersion in the vertical direction (distance)

u = mean wind speed (distance/time)

pi = 3.141593

For the purpose of a worst case assessment, it can be assumed that an individual is standing on a down wind edge of the Job Corps Landfill, and is exposed to PCB vapors emitted from an area of 45 m x 1 m (the landfill is approximately 0.5 acre in area, or about 45m x 45m). Thus, the PCB emission rate is 45 m^2 x 1.13 ug/sec/m^2 or 50 ug/sec. Using this value to solve for C(x).

	Case	u (m/sec)	Y	Z	C(x) ug/m3
1	1 hr	1	1	2.5	6.37
2	24 hr	2	2	3.5	1.14
3	7 day	3	6	5	0.18

The relative contribution of this component of airborne exposure is discussed below.

For the purposes of this analysis, it is assumed that an adult male (70 kg) is exposed to an average of 10 mg/m3 of respirable dusts and soil particles (a level considered to constitute a nuisance) kicked up during a 4-hour recreational activity at the site. The soil monitoring data at this site gave a geometric mean value of approximately 7,950 mg/kg dry weight of total PCBs in soil, with a maximum of 50,000 mg/kg wet weight (69,042 mg/kg dry weight). Also assuming an inhalation rate of 1.3 m3/hour for light activity (USEPA ECAO-CIN-477, 1985) and that all the inhaled PCB is bioavailable, a total exposure of 413 ug PCB is obtained in the 4 hour period using the mean PCB concentration, or 5.9 ug/kg body weight. In comparison, a four hour exposure to vapors of PCBs under worst case conditions as calculated using the air model above would produce a total exposure of 1.3 m3/hr x 6.37 ug/m3, or 33 ug, less than 8 percent of the component of exposure from dust inhalation. For this reason, exposure to PCBs in the vapor state was not considered further in this assessment. Assuming three 4-hour visits per year, a daily inhalation intake of 0.05 ug/kg/day is estimated.

Comparable exposure by inhalation to soil residues of N-nitrosodimethylamine at 440 ug/kg results in a daily intake of 2.6×10^{-6} ug/kg/day. The value for lead assuming the same exposure scenario and a soil level of 5,000 mg/kg would be 0.03 ug/kg/day.

The acute toxicity of PCBs and lead is quite low to humans and terrestrial animals. For instance, the acute oral LD50 of Aroclor 1254 in adult Sherman rats is 4 to 10 g/kg body weight (USEPA AWQC, 1980). Acute inhalation toxicity data were not located, but an intravenous LD50 value in the rat of 0.358 g/kg for Aroclor 1254 may be a worst case ap-

proximation. Thus, a safety margin of approximately 100 or more exists for acute inhalation exposure to PCBs of a human under reasonable worst case conditions. Lead will be considered in Section 24.4.2.2.

Because the Job Corps landfill is very rarely used for recreational purposes, chronic exposures at the site do not exist for humans. The contribution of inhaled soil particles containing PCBs and lead to lifetime health risks in humans is discussed in Section 24.4.2.2.

Unlike humans, chronic exposures of wildlife to PCBs and lead at the site are likely by inhalation and other routes. The site provides good habitat for small mammals such as mice, chipmunks, and the like which may inhale soil dusts and vapors during burrowing activities. As with humans, no models were located for estimating inhalation exposures in small mammals under these field conditions. Using an active breathing rate value of 0.0042 m3/hour for a 30 g mouse (USEPA ECAO-CIN-477, 1985) and creation of a 10 mg/m3 dust of 7,950 mg/kg dry weight PCBs during 1 hour of daily burrowing, a daily chronic inhalation exposure of 0.016 mg/kg body weight PCB is obtained. The comparable value for lead at a soil level of 5,000 mg/kg is 0.004 mg/kg. In addition, all rodents living in burrows within the contaminated area of the landfill (within the top 1 foot of surface soil) would be expected to inhale a significant portion of PCB vapors enumerating from the soil. Cadmium or lead vapors are not expected to be significant due to the affinity of these metals for soils and the low vapor pressures of these metals compounds. Assuming that the air is saturated with PCB vapors at the present conditions at the landfill, a still wind speed of 1 mph, and an estimated residence time of 16 hours in the burrow (one hour of which is spent burrowing, the remainder resting), an exposure of 1.03 mg/kg day is

estimated for this route. Table 24.1 presents additional detail for the wildlife exposures estimated for this site.

Other types of wildlife which might receive exposures via the inhalation route at this site include deer, rabbit, mink and otter; the corresponding exposure levels for each species depends on their breathing rates, body weight and habitat. The magnitude, contribution and significance of these airborne dusts to total lifetime exposure are presented in Section 24.4.2.2, Quantitative Risk Assessment Table 24-1.

Estimates of Exposures by Direct Contact

The direct contact route of exposure at the Job Corps landfill has been identified as complete for both humans and wildlife. However, since PCBs and lead are very tightly bound to soils and sediments, dermal absorption of PCBs and lead is not expected to be significant for humans. The contributions from this exposure pathway for wildlife are addressed below in the evaluation of ingestion exposures, as a result of prolonged intimate contact with contaminants in soil during daily burrowing activities. Rather, the pathway consists of ingestion of soil-bound residues picked up through direct contact with the soils. Therefore, the contribution of this route of exposure will be discussed in the following section on ingestion exposures.

Estimates of Ground Water Exposures

As previously discussed in Section 24.4.1.2, groundwater residues of PCBs and lead are minimal in the silty soils encountered at this site, and most likely resulted from the fill material during the drilling of monitoring wells. Furthermore, there are no ground water users at this site or in

the eastern portion of the Refuge. Therefore, the ground water exposure pathway is incomplete, and will not be considered quantitatively.

Estimates of Exposure by Surface Water

The qualitative assessment has identified the surface water route as complete at the Job Corps site in view of the direct contact of the landfill with the pond, the presence of PCBs and other waste components in the pond water and sediments, and the availability of an outlet stream for offsite transport of residues. Although the pond may serve as a major source of drinking water for large mammals (i.e. deer, raccoons, beaver) and waterfowl, it will not produce significant human exposures.

Surface water ingestion by wildlife will be considered in the following section on ingestion. The presence of PCB residues in the pond sediments and water column provide a source of chronic exposure by aquatic organisms and water fowl. Mean values of total PCB residues were calculated from site monitoring data to be 0.045 ug/L for pond water and 109,635 ug/kg for sediment. USEPA AWQC (1980) has established a criterion of 0.014 ug/L of PCB to be protective of freshwater aquatic life under chronic exposure conditions and 2.0 ug/L under acute exposure conditions. Thus, pond water concentrations of PCBs exceed the AWQC chronic toxicity criteria for freshwater aquatic life, but not the acute exposure level.

While there are probably too few data points to currently establish an accurate mean concentration of PCBs in the pond water, resident organisms will also receive exposure via contact with contaminated bottom sediments and ingestion of residues accumulated in food chains from the sediments. It is feasible that populations in the pond may be at risk

from exposure to PCB residues in bottom sediments, although a limited survey of pond fish did not show detectable PCB levels, and a qualitative survey of pond life (Section 24.4.1.2) would suggest that such impact is not grossly obvious.

Estimates of Ingestion Exposures

Human

PCBs as a class are highly lipophilic materials which are readily taken into the body and tend to resist metabolic destruction and elimination. Thus, the well documented ability of PCBs to bioconcentrate in organisms from ambient media and food is one of the most significant chemical and toxicological properties of these materials. The presence of PCB residues in surface soils and in the pond water and sediments presents a source of exposure via ingestion of accumulated residues of food items by terrestrial and aquatic wildlife.

As discussed in the Receptor Evaluation, human consumption of fish caught in the Job Corps Pond will be negligible, since recreational fishing at this site is not permitted, and nearby Crab Orchard Lake offers an abundance of popular game fish for the sports fishermen. In addition, a limited survey of pond fish did not show detectable level of PCBs (0.4 mg/kg detection limit). This pathway is therefore technically incomplete for humans and ingestion of fish will not be considered in the quantitative assessment of human exposures.

Human ingestion exposures to PCBs may also occur by consumption of contaminated venison taken during the limited Refuge deer season. However, as cited in Section 2.6.4 of this report, analysis of ten Refuge deer for PCB contamination showed no detectable residues in either fat or

red meat deer tissue. Given these data and the transient nature of grazing by deer at the site, the probability of human exposure appears nil.

For purposes of the evaluation of ingestion exposure, an individual is assumed as a worst case estimate to consume 100 mg of soil contaminated with 7,950 mg/kg of PCBs per trip to the site (i.e. for hunting), as the result of direct contact with the soil to skin, boots and clothing. This produces a total ingestion exposure of 0.011 mg/kg/trip for a 70 kg adult. For the worst case assumption of three such trips per year, recurring over a 70-year lifetime, a daily intake of 0.093 ug/kg/day is estimated for the ingestion route (see Table 24-1). A comparable value for lead exposure would be 0.058 ug/kg/day for ingestion of soil containing 5,000 mg/kg lead. The corresponding daily intake from ingestion of soil containing N-nitrosodimethylamine at 440 ug/kg would be 5.2×10^{-6} ug/kg/day. The potential human health risks associated with these chronic exposures are discussed in Section 24.4.2.2.

Wildlife

Data are limited for estimating the chronic effects of PCB and lead to wildlife. USEPA AWQC (1980) discussed studies indicating mortality and reproductive failure of mink fed PCB-adulterated fish meal. A nine month feeding of 2 mg/kg dietary Aroclor 1254 significantly inhibited reproduction and 10 mg/kg resulted in complete mortality of pregnant female mink. For purposes of the wildlife assessment, it is assumed that PCB concentrations in fish are present at one-half the detection limit used in the survey of pond fish (since the analyses did not measure detectable residues), for a concentration of 0.2 mg/kg. An assessment of the exposures to wildlife from pond fish is discussed in Section 24.2.2.2.

Wildlife may also receive exposure from inadvertent ingestion of contaminated soil, vegetation, or sediment via preening or burrowing activities and consumption of food at the site. The magnitude of such exposures is also a function of food preference, home range, and migratory pattern of the species. For instance, deer move relatively long distances (large home range), feed on or browse through vegetation located higher off the ground and do little preening. Rabbits have a small home range, feed on broad leaf vegetation located close to the ground that usually has higher contaminant concentrations due to rain splash, and they preen regularly. Some duck (diving) species dabble in sediments but only spend part of the year in an area and when settled they move frequently from one body of water to another. Such species would not be expected to receive significant exposures via residues in the pond sediments since these might spend approximately two months per year at the Refuge, and during this time would reside preferentially in nearby Crab Orchard Lake. On the other hand, duck species which might remain on-site long enough to receive any quantifiable exposures are predominantly vegetarian types such as mallards or other surface dwellers, and could be exposed to contaminants as a result of feeding on terrestrial and aquatic vegetation. Mice live close to the soil, preen regularly, and may consume roots or other plant parts growing close to the soil that can accumulate relatively high levels of contaminants from rain drop splash and wind erosion. In order to assess the potential risks to wildlife posed by site soil and vegetation residues, these species were chosen to represent a variety of such biological and ecological variables. The magnitude of these ingestion intakes of soil, sediment and vegetation

and their significance in a lifetime of exposure to site PCB residues is detailed in the following section on Quantitative Risk Assessment.

Exposure of small burrowing animals such as mice to soil residues is also possible by dermal absorption, as well as by direct ingestion. No guidance was located for conducting an exposure assessment of such an event. Key elements of such an assessment, such as the available body surface area of a generally fur-covered animal through which such absorption might occur and the rate of dermal transfer of soil bound PCBs are a matter of conjecture. Realizing these uncertainties, the following scenario was constructed.

It is assumed that a small burrowing rodent such as a 30 g mouse lives in intimate contact with site soils containing a level of 7,950 mg/kg PCBs. It is further assumed that the body surface area available for absorption includes close to the entire body (e.g. for species that dust bathe), and that this area is about 36 cm^2. Due to the animal's grooming habits, it is assumed that soil contacting the fur of the animal is inadvertently consumed, and that these residues are considered under the direct contact/ingestion scenario discussed above. Hawley (1985) assumed that soil dust would adhere to human skin at a rate of 3.5 mg/cm^2. Assuming a similar rate for the mouse, the exposed skin of the animal will be in continuous contact with 126 mg of soil containing 7,950 mg/kg. As reviewed in USEPA 600/6-86/002, Development of Advisroy Levels for PCB cleanup (1987), an absorption fraction of 0.05 might be used for Aroclor 1254. Assuming that the soil residues of PCBs are absorbed at this rate every 24 hour period of exposure, it can be estimated that the mouse receives an exposure to soil PCBs at a rate of 1.67 mg/kg/day.

Wildlife may also be exposed to low level PCBs via drinking the pond water. The quantitative assessment for ingestion of pond water is detailed in Section 24.4.2.2, based on a mean PCB concentration of 0.045 ug/L in pond waters, and assuming a water consumption rate of 10% of body weight per day for herbivores, and a rate of 30% of body weight for carnivores (Chew, 1965).

The pond outlet stream provides a mechanism for offsite transport of PCBs via dissolved and sediment-bound residues. However, insufficient data on downstream residue levels and potential receptors prevents exposure and risk estimates. Dilution effects should lessen the downstream exposures relative to the worst case pond situation.

Burrowing animals such as mice are estimated to ingest 1.5 grams as food (10% soil, 90% vegetation), based on a consumption of 5% of body weight. Soils at this site contain an average of 5,000 mg/kg lead, while lead levels in vegetation are assumed as 1 percent of the soil concentration, based on the fraction expected to leach from soils for uptake by site plants. An exposure of 47.5 mg/kg/day of lead is estimated as a result of daily burrowing and inadvertent ingestion of dust while grooming. Heavy metals may also bioaccumulate in organisms such as earthworms which ingest soil while feeding. In particular, cadmium has been demonstrated to accumulate in earthworms by a factor of 50-fold over soil concentrations (Ruelle, 1987). This creates the potential for ingestion exposures in wildlife via contamination of food chains, as well as the pond. The significance of these exposures is discussed in the following section.

24.4.2.2 Quantitative Risk Assessment

The estimates of total PCB intakes of receptors in complete exposure pathways are summarized below:

Human Risks

Human PCB exposure at the Job Corps site is technically acute in nature rather than chronic and is much lower than any which would raise concerns for acute PCB toxicity. For instance, access to the Refuge for deer hunting is granted on a limited basis, greatly reducing the probability for repeated, chronic exposures as assumed in the following worst case estimate. The chronic PCB toxic effect of most concern is carcinogenicity as evidenced by lifetime dietary studies in rodents. The risk of excess cancer caused by human direct contact exposures at the Job Corps site may be estimated by multiplying the average daily intake of PCBs estimated in Table 24-1 by the PCB unit cancer risk factor (i.e. potency factor; a dose-response factor derived from a conservative mathematical extrapolation of the animal cancer data). For PCBs, the unit cancer risk factor determined by EPA is 7.7 $(mg/kg/day)^{-1}$ (Exhibit A). Therefore, the cancer risk estimate for humans due to PCB exposure is $7.7 \times 1.46 \times 10^{-4} = 1.1 \times 10^{-3}$.

An additional potentially carcinogenic substance, N-nitrosodimethylamine, was also present at this site. However, the quantitative data for this compound is based only on a single detection in soil during the Phase I investigation. Current risk assessment methodology suggests that risks from concurrent exposures to a mixture of carcinogenic substances can be approximated by addition of the risk estimates of the individual substances. The exposure assessment determined that, under the assumed exposure conditions, using twice the level detected at the site, N-nitrosodimethylamine residues in the soil might result in a daily intake

TABLE 24-1 (p. 1)

SECTION 24.4.2: JOB CORPS LANDFILL & POND
QUANTITATIVE RISK ASSESSMENT

SPECIES	BODY WEIGHT kg	ESTIMATED DAILY PCB INTAKE												
		Inhalation Rate (1)				Food (2)			-Pond Water (3)-		-Pond Fish (4)-		-Dermal (5)-	TOTAL
		air m3/hr	dust ug PCB	vapor ug PCB	PCB ug/kg	Soil grams	Veget/ Insects grams	PCB ug/kg	Liters	PCB ug/kg	Fish grams	PCB ug/kg	Absorption mg/kg PCB	EXPOSURE mg/kg/day
MAN	70	1.3	3.40	0.27	0.05	0.10	~0	0.093	~0	~0	~0	~0	~0	1.46E-04
Deer	60	1.3	413	33	7.44	170	1530	24,552	6	0.270	-	-	~0	24.56
Mallard	1	-	~0	~0	~0	5	45	42,523	0.1	0.0045	~0	~0	~0	42.52
Rabbit	1	0.083	6.63	~0	6.63	7.5	67.5	64,991	0.1	0.0045	-	-	0.01	65.01
Mouse	0.03	0.006#	0.48	33	1,104	0.15	1.35	43,328	0.003	0.000135	-	-	1.67	46.10
Mink	1	0.083	6.63	~0	6.63	~0	75	5,963	0.3	0.0135	75	15	~0	5.98
Heron	3	0.25	20	~0	6.63	~0	~0	~0	0.9	0.0405	350	23	~0	0.030
Otter	9	0.75	60	~0	6.63	~0	~0	~0	2.7	0.1215	900	20	~0	0.027

~0 = negligible contribution
(-) Not applicable or irrelevant
(#) Active breathing rate used for dust exposure. Resting rate of 0.0015 m3/hr used for vapor exposure.
Additional assumptions listed on Page 2.

TABLE 24-1 (p. 2)

ASSUMPTIONS

(1) INHALATION EXPOSURE

Dust Inhalation: Based on 10 mg dust inhaled per m3 of air, 7.95 ug mean PCBs per mg dust, and 1 hr (small mammals) or 4 hrs (man and deer) exposure duration each contact.

Vapor Inhalation: Exposures for man and deer estimated based on air model by Mwang, et.al. (1986); see text. Vapor exposures for burrowing animals based on saturation concentration of Aroclor 1254 (vapor pressure = 1.49E-06 psia) resulting in approx. 1.36 mg PCB/m3 air, species breathing rate, and estimated duration in burrow (16 hrs for mouse).

(2) INGESTION EXPOSURE

Total Food (Vegetation + Soil + Terrestrial mammals/insects):
Intake for wildlife calculated as 5% of body weight, except deer (1.7 kg/day, Vulmey and Ulrey, 1982), rabbit (75 g/day, Green and Dunsmore, 1978), and mink (150 g/day, 50% as fish (Toweill and Tabor, 1982; G. Smith, 1988), and 50% as terrestrial insects/mammals (Gerould, 1988). Diet for mallard reflects preference of 75% terrestrial insects and 25% vegetation in consideration of one of the most susceptible periods of their life cycle. Mink and mallard total food exposures are adjusted for home range; the affected area is estimated to comprise 10% of the home range for these species.

Soil/Sediment: Ingestion rate based on mean PCB concentration of 7.95 ug/mg soil and inadvertent consumption of particulates while preening or burrowing (wildlife), or ingested as a result of soiled clothing or boots (man).

Vegetation Concentration of PCBs estimated as 1% of mean soil concentration or 7.95 mg/kg.

Terrestrial Animals PCB concentration assumed to be 10% of residual soil concentration (i.e. 795 mg/kg). (Gerould, 1988).

(3) Water intake rates assumed to be 10% of body weight for hervibores, 30% of body weight for carnivores, and a mean pond water concentration of 0.045 ug/L.

(4) Fish and aquatic foods are assumed to contain PCB residues at one half the analytical detection limit based on a limited survey of fish which did not show detectable PCB residues (0.4 mg/kg detection limit). Mink may consume 50% or 75 g/day of their total diet as fish (Toweill and Tabor, 1982; G. Smith, 1988). Otter are estimated to consume 900 g fish/day (90% of 1 kg food/day, Chapman and Feldhammer, 1982); great blue herons consume 350 g food/day (Kendeigh, 1970; Kushman, 1977) consisting mostly of fish and aquatic organisms.

(5) DERMAL ABSORPTION

Dermal absorption exposure would be negligible for transient, large terrestrial mammals such as humans or deer. Absorption exposure would be negligible for species which bathe frequently in water, or otherwise spend little time in direct contact with soil (e.g. mink, heron, otter, mallard). Exposure estimates for rabbit and mouse are based on approx. body surface area available for constant contact (10 cm2 for rabbit and 36 cm2 for mouse), adhesion of 3.5 mg soil per cm2 of body (Hawley, 1985), and 5% extent of absorption for Aroclor 1254 (EPA/ORD, 1986).

of 5.2×10^{-6} ug/kg/day. USEPA (1987) estimated that a daily intake of 0.0137 ug of this compound may be associated with a risk of excess cancer of 10^{-5}. Thus, assuming that the 440 ug/kg soil residue of nitrosamines is representative of the entire site, a cancer risk of 2×10^{-7} may be posed by site exposures to N-nitrosodimethylamine. Adding this risk to the risk estimated for PCB exposure at the site, a total carcinogenic risk of 1.1×10^{-3} is estimated.

This level of risk is higher than the 10^{-7} to 10^{-4} population risk level generally considered by regulatory and public health agencies to pose minimal excess health concerns on a national basis. This estimate used very conservative values which are mitigated by several major factors discussed in Section 24.4.3, Analysis of Uncertainties.

Total human intake of lead was estimated to be 0.088 ug/kg/day for this scenario. For comparative purposes, the USEPA OHEA (1987) has determined an acceptable daily intake of lead to be 1.43 ug/kg/day. Thus, the estimated upper bound lead exposure possible at this site is well below the acceptable daily intake level of 1.43 ug/kg/day.

Cadmium is a human carcinogen by the inhalation route and produces serious harm to the kidney at low levels of chronic ingestion. Its carcinogenic potency is estimated to be greater than that of PCBs (Exhibit A). Therefore, chronic exposure to soil residues of cadmium would be of great concern if it were to be shown that it was present at consistently high levels at this site.

Wildlife Risks

The mean water concentration of PCB in the pond is below the criterion of 2 ug/L considered by USEPA AWQC (1980) to protect

TABLE 24-2 (p. 1)

JOB CORPS LANDFILL & POND
EXPOSURE ASSESSMENT FOR 50 MG/KG CLEANUP

SPECIES	BODY WEIGHT kg	ESTIMATED DAILY PCB INTAKE												
		Inhalation Rate (1)				Food (2)			Pond Water (3)		Pond Fish (4)		Dermal (5)	TOTAL
		air m3/hr	dust ug PCB	vapor ug PCB	PCB ug/kg	Soil grams	Veget/ Insects grams	PCB ug/kg	Liters	PCB ug/kg	Fish grams	PCB ug/kg	Absorption mg/kg PCB	EXPOSURE mg/kg/day
MAN	70	1.3	0.02	0.27	0.0042	0.10	~0	0.001	~0	-	~0	~0	~0	4.77E-06
Deer	60	1.3	3	33	0.593	170	1530	154	6	-	-	-	~0	0.155
Mallard	1	-	~0	~0	~0	5	45	267	0.1	-	~0	~0	~0	0.267
Rabbit	1	0.083	0.04	~0	0.04	7.5	67.5	409	0.1	-	-	-	0.0001	0.409
Mouse	0.03	0.006#	0.003	7.20	240	0.15	1.35	273	0.003	-	-	-	0.011	0.523
Mink	1	0.083	0.04	~0	0	~0	75	37.5	0.3	-	75	15	~0	0.053
Heron	3	0.25	0.13	~0	0.042	~0	~0	~0	0.9	-	350	23	~0	0.023
Otter	9	0.75	0.38	~0	0.04	~0	~0	~0	2.7	-	900	20	~0	0.020

~0 = negligible contribution
(-) Not applicable or irrelevant
(#) Active breathing rate used for dust exposure. Resting rate of 0.0015 m3/hr used for vapor exposure.
Additional assumptions (notes) listed on Page 2.

TABLE 24-2 (p. 2)

ASSUMPTIONS

(1) INHALATION EXPOSURE

Dust Inhalation: Based on 10 mg dust inhaled per m3 of air, 0.05 ug mean PCBs per mg dust, and 1 hr (small mammals) or 4 hrs (man and deer) exposure duration each contact.

Vapor Inhalation: Exposures for man and deer estimated based on air model by Kwang, et.al. (1986); see text. Vapor exposures for burrowing animals based on emission rate of 0.61 ug/m3 at 1 mg Aroclor 1254/kg and 10 mph wind, corrected for 50 mg/kg mean concentration and low air turnover within burrow space (1 mph wind), for a modified emission rate of 305 ug/m3. Estimated duration in burrow 16 hrs (for burrowing rodent).

(2) INGESTION EXPOSURE

Total Food (Vegetation + Soil + Terrestrial mammals/insects):
Intakes for wildlife calculated as 5% of body weight, except deer (1.7 kg/day, Mulmey and Ulrey, 1982), rabbit (75 g/day, Green and Dunsmore, 1978), and mink (150 g/day, 50% as fish (Towell and Tabor, 1982; G. Smith, 1988), and 50% as terrestrial insects/mammals (Gerould, 1988). Diet for mallard reflects preference of 75% terrestrial insects and 25% vegetation in consideration of one of the most susceptible periods of their life cycle. Mink and mallard total food exposures are adjusted for home range; the affected area is estimated to comprise 10% of the home range for these species.

Soil/Sediment: Ingestion rate based on mean PCB concentration of 0.05 ug/mg soil and inadvertent consumption of particulates while preening or burrowing (wildlife), or ingested as a result of soiled clothing or boots (man).

Vegetation Concentration of PCBs estimated as 1% of mean soil concentration or 0.5 mg/kg.

Terrestrial Insects PCB concentration assumed to be 10% of residual soil concentration (i.e. 5 mg/kg). (Gerould, 1988)

(3) Residual PCB levels in the pond water would be non-detectable following remediation. Due to the lack of a waste source, this pathway is technically incomplete.

(4) Fish and aquatic foods are assumed to contain PCB residues at one half the analytical detection limit based on a limited survey of fish which did not show detectable PCB residues (0.4 mg/kg detection limit). Mink may consume 50% or 75 g/day of their total diet as fish (Towell and Tabor, 1982; G. Smith, 1988). Otter are estimated to consume 900 g fish/day (90% of 1 kg food/day, Chapman and Feldhamer, 1982); great blue herons consume 350 g food/day (Kendeigh, 1970; Kushnan, 1977) consisting mostly of fish and aquatic organisms.

(5) DERMAL ABSORPTION

Dermal absorption exposure would be negligible for transient, large terrestrial mammals such as humans or deer. Absorption exposure would be negligible for species which bathe frequently in water, or otherwise spend little time in direct contact with soil (e.g. mink, heron, otter, mallard). Exposure estimates for rabbit and mouse are based on approx. body surface area available for constant contact (10 cm2 for rabbit and 36 cm2 for mouse), adhesion of 3.5 mg soil per cm2 of body (Hawley, 1985), and 5% extent of absorption for Aroclor 1254 (EPA/ORD, 1986).

freshwater aquatic species from acute toxicity, although high PCB levels in the sediments may pose an undefined acute toxicity risk to benthic organisms. However, the ambient water quality criteria for chronic effects in freshwater aquatic organisms (0.014 ug/L) is exceeded by at least three-fold in the Job Corps Pond. While a qualitative examination of the pond showed no gross evidence of ecosystem impairment, it is possible that sensitive elements of the pond community are being affected by chronic PCB exposure.

Exposures to PCBs for several typical terrestrial wildlife species were evaluated for the site. For comparative purposes, a 1.0 kg adult female mink consuming 150 g daily of a 2 mg/kg PCB diet receives a daily intake of 300 ug/kg/day which is a chronic effect level for this species. The mink, however, may be unusually sensitive to PCB effects relative to other mammals.

The most obviously affected species may be fish-eating mammals (mink, otter, raccoon) and birds (herons, merganser ducks, osprey), even though a limited survey of Job Corps pond fish did not detect PCBs. If these species showed the same sensitivity to PCBs as shown by mink and chickens in laboratory studies, reproductive failure and possibly overt lethality may occur at this site. The estimated whole fish PCB concentration of 12.5 mg/kg could potentially exceed the chronic dietary levels (2-5 mg/kg) producing lethality and reproductive failure in mink, a piscivorous mammal (USEPA AWQC, 1980). Even using less conservative estimates for BCF and water PCB concentrations, sensitive piscivorous mammals such as mink may be at risk from pond PCB contaminants, provided they accumulated to the levels estimated based on their physical properties.

Fish-eating birds such as herons and certain ducks may also be at risk but insufficient data exist to estimate degree of risk. Of particular concern is the location of two active bald eagle (an endangered piscivorous species) nests on the Refuge, since fish from the site pond might be a source of prey (Ruelle, 1987). Domestic chickens given 20 mg/kg dietary PCB displayed a broad spectrum of reproductive and teratogenic effects (Exhibit A). A quantitative assessment of exposures to bald eagle due to ingestion of PCB residues in fish is presented in Section 38.4, Wildlife Assessment for Crab Orchard Lake. No PCBs were detected in fish sampled at the Job Corps Pond, and, due to the large foraging range of these species, exposures to bald eagles at this would not be expected to be significant. For other species such as deer and duck, incidental, short term exposure to pond sediments while browsing or searching for food are relatively low based on the site conditions and habitat (see Table 24-1).

USEPA ECAO-CIN-414 (1987) determined that a daily exposure of rats to 1 mg/kg/day of Aroclor 1254 constituted a subchronic no adverse effect level in that species, compared to the site ingestion exposure estimate for mice of 46 mg/kg/day. Rabbits are estimated to receive total exposures on the order of 65 mg/kg/day at this site. Thus, small herbivorous mammals such as mice and rabbits are likely to be at risk from site PCB residues.

An alternate assessment of wildlife risks from exposure to PCBs was developed independently by the U.S. Fish and Wildlife Service, which is presented in its entirety in Exhibit D. Using a three-phase fugacity model and dietary/dermal exposure assumptions, total daily PCB exposure was estimated for a burrowing animal such as a pocket gopher. As

developed, approximately 90 percent of the total exposure is determined by inhalation of PCBs in the animal's burrow. The air concentration is related to soil PCB concentration using a partitioning model based on the estimated fugacities of PCBs in air, soil, and water, as developed by Mackay and colleagues (Mackay, 1985).

It is assumed that the exposed animal spends all of its time in a sealed burrow, actively burrowing 9 hours per day and resting 15 hours per day. It is assumed that the PCB concentrations in air, soil, and water are in thermodynamic equilibrium at all times. Soil concentrations were back-calculated to represent concentrations which would result in exposures comparable to food consumption rates employed in laboratory tests for wild and domestic rodent species, which were associated with biological and toxicological effects. Two examples are presented:

1) 0.7 mg/kg soil PCBs would result in exposures comparable to the dietary PCB exposures in an experiment which was found to "... increase the liver weights in F1 male weanling rats..." and to "... decrease the circulating levels of adrenal cortex hormone B...".

2) 7 mg/kg to 14 mg/kg soil PCBs would result in exposures comparable to the dietary PCB exposures in an experiment which was found to "... decrease the weight of reproductive organs, growth rates, and reproductive success of second generation white-footed mice...".

A soil sample analyzed in the Phase I survey of the Job Corps site showed a trace level (0.22 mg/kg wet weight) of N-nitrosodimethylamine. In order to provide a conservative estimate of the potential effects from

this contaminant, and due to deficiencies noted in the analytical result for this soil sample, the quantitative evaluation for humans as well as for wildlife is based on twice the detected level or 440 ug N-nitrosodimethylamine per kg of exposed soil. Because of the intimate contact which small burrowing mammals may have with soil, the risks of direct contact of wildlife to these residue levels are assessed below, based on available information on the effects of nitrosoamines summarized in the risk assessment for Site 19 (see Section 26.4).

Using the assumptions given above for exposures of burrowing mice (30 g body weight) to site PCB and lead residues, exposure to soil residues and vegetation (assumed to contain 1% of the soil concentration) of 0.44 mg/kg N-nitrosodimethylamine will produce a daily intake of 4.2×10^{-3} mg/kg/day by the ingestion route. Vapor inhalation would not be expected to be significant at the low microgram level of contaminant observed. Inhalation of contaminated dusts (e.g. during burrowing) are estimated at 8.8×10^{-7} mg/kg/day. Using the unit risk factor of 26 $(mg/kg/day)^{-1}$ derived in USEPA ECAO (1986--) based on rat studies, and assuming a similar sensitivity for wild rodent species, a cancer risk estimate of 1×10^{-1} is derived for the total potential exposure received by burrowing rodents at this site. As discussed in Section 26.4, the significance of this risk level is uncertain. The effects of cancer, generally incurred later in an exposed organism's lifetime, may be very small when considered in light of other factors influencing whether a wildlife population can maintain itself (i.e. survival to reproductive age, competition, weather, disease, predation, etc). On this basis, Newell et al. (1987) chose a risk level of 10^{-2} as a level of acceptable carcinogenic risk for wildlife, with the acknowledgement that more study is needed to

justify this choice. Using this rationale, it is concluded that wildlife exposure to site residues of N-nitrosodimethylamine could result in a carcinogenic response, but the biological significance of the response cannot be assessed with currently available information.

Additional studies were reviewed to further evaluate wildlife exposures to N-nitrosodimethylamine at this site. USEPA ECAO (1986--) used a reproductive effects study in adult female mice to establish a lowest observed effect level (LOEL) of 0.019 mg/kg/day for N-nitrosodimethylamine. The estimated daily exposure rate of 0.0042 mg/kg/day at this site is below this LOEL. Thus assuming exposed species are of equivalent sensitivity to N-nitrosodimethylamine as laboratory mouse strains, the estimated exposure rates are below a level which may elicit a toxic response under subchronic conditions.

As estimated in the previous section, burrowing mammals and other site wildlife may be endangered by chronic exposure to site lead levels of 47.5 mg/kg/day. The effects of these lead residues on wildlife species are largely undocumented, but using the human criterion for acceptable intake levels, wildlife ingestion of even a small fraction of this value may produce reproductive impairment and possibly other subtle effects which might decrease survival of wild populations. A similar argument may hold for aquatic organisms and piscivorous species exposed to lead via aquatic foodchains, although insufficient data exists to assess an exposure/response relationship.

Cadmium is also potentially toxic to fish and wildlife, and can biomagnify to potentially toxic levels in soil-ingestion foodchain animals such as earthworms. Small predaceous birds and mammals would be at particular risk if cadmium was consistently elevated at this site.

24.4.3 Analysis of Uncertainties

As discussed in Section 6.5, most approaches to quantitative risk assessment are inherently conservative in order to be most protective of public and environmental health in the face of numerous scientific uncertainties and insufficiencies in case-specific data. At the Job Corps landfill site, worst case estimates indicated that risks to environmental populations may be posed by PCBs and nitrosoamines residues at the site, particularly by inhalation and ingestion of dust-borne residues. Several areas of uncertainty exist, however, which may serve to mitigate the quantitative degree of risk. First, the level of contaminants in fish consumed by wildlife were estimated using reasonable worst case assumptions on bioconcentration potential. The estimates of exposure to wildlife due to ingestion of fish were based on an average potential bioconcentration factor, although a limited survey of pond fish did not show detectable PCB levels. The fish data were used only qualitatively to support the conservative nature of the wildlife assessment, since only two samples were collected following the Phase II sampling effort.

The worst case human cancer risk assessment for PCBs and for chronic toxicity from lead was conducted using an assumption of lifetime visits (i.e. for hunting), a highly improbable event. It is not likely that the same individual will continue to hunt on this site annually, three times each year over an entire 70-year lifetime. The assessment developed for N-nitrosodimethylamine was based on doubling the result from a single soil sample analyzed in the Phase I survey, although the purpose of that survey was to screen the site and not to support a quantitative assessment. Thus, the limited analytical data for nitrosoamines do not provide for a confident estimate of the risks associated with nitrosamines

exposure. Furthermore, the quantitative assessment model that was used assumes that there is no exposure level for PCBs which does not pose a risk from cancer. However, there is accumulating evidence that PCBs may induce cancer in animal tests through a threshold mechanism, such as promotion of pre-existing lesions (Williams and Weisburger, 1986). Thus, it is possible that low exposures to PCBs may not present concerns for carcinogenic effects.

Finally, the size of the potentially exposed human population is at best very small and it is unlikely that an increased occurrence of cancer at even a 10^{-3} risk level could be detected above background levels of disease, since the probability that even one individual would meet the worst case exposure criterion is almost nonexistent. Assessment of human and wildlife risks from lead exposure is limited by lack of knowledge of the types of insoluble lead salts (i.e. sulfate, oxides) likely to be present at the site. It has been speculated that the geese found in 1985 at this site (See Section 24.1) may have died as a consequence of contaminants encountered here. However, given the limited data base, it is not appropriate to implicate site contamination as the reason for geese kills at the site, since no PCBs or other organic were detected in the carcasses and metals were not analyzed. It is of interest to note that some of the better documented cases of wildlife lead toxicity concern waterfowl (Exhibit A) presumably due to ingestion of lead shot while feeding in aquatic environments subject to hunting.

24.5 Preliminary Remedial Alternatives

On the basis of the above assessment, it was determined that the levels of contaminants present pose risk levels to exposed human and

wildlife receptors. The major contaminants of concern in soils, sediments and waters were identified to be cadmium, lead and PCBs. The explosive residue nitrobenzene was also detected in both ground water and surface water at levels below the AWQC. The contaminants in soils were found to be associated with only the top 0-1 ft. of soil in the landfill, while subsurface soils (between 1 and 3 ft. depth) did not contain detectable concentrations of contaminants with the exception of one core which showed 11.6 mg/kg PCBs and 219 mg/kg lead. Contamination in sediments was detected in the surface samples (109.6 mg/kg wet wt PCBs to 1 ft depth), but no subsurface sediments were collected to verify the levels in subsurface sediments. Fish samples from the pond showed lead levels up to 6.9 mg/kg (bluegill only), but PCBs were undetected (0.4 mg/kg detection level) and only traces of cadmium and mercury were found.

In its current condition, the Job Corps Site should not be used for any activities which would increase the potential for human and/or wildlife exposure via the water or direct contact with soils/sediments. Future uses of the site might be considered, subject to additional testing to assure the site does not constitute a risk to potential human or wildlife receptors.

Risk levels from site contaminants may be reduced to a range of 10^{-6} to 10^{-5}, generally considered acceptable by regulatory and health agencies, by implementation of appropriate remedial actions. In Section 24.4.2.2, it was estimated that PCB soil concentrations of 7950 mg/kg presented an excess cancer risk of 1.1×10^{-3} for the assumed scenarios of exposure. Using similar assumptions, the risks to humans would be reduced to an acceptable range of 10^{-6} to 10^{-5} by precluding exposure to soils containing PCB concentrations greater than 7 to 70 mg/kg. Section

24.4.2.2 also indicates that the most sensitive wildlife species, mink, shows reproductive lowest observed effects with chronic exposure to PCBs at 640 ug/kg/day (Newell, 1987). The 7 to 70 mg/kg level of remediation of exposed surface soils is 14 to 1.4 fold lower than the no observed effect level for protection of the most sensitive wildlife species. Exposure estimates under an example cleanup scenario of 50 mg/kg PCBs in soil are presented in Table 24-2. The assumptions used in calculating such exposures are similar to those used in the site risk assessment (Section 24.4).

In general, the objectives of the remedial program to be developed in the FS will be to render incomplete all possible transport routes between contaminant sources of concern and potential receptors, including the routes of direct contact (absorption, ingestion), surface water transport, and inhalation of contaminated dusts or vapors. With respect to contaminated transport pathways such as drainage ditches (as opposed to contamination sources) identified as a concern, such remedial measures as surface excavation, capping, regrading, revegetating, and surface water diversion will be emphasized in the FS.

Table 2 of the Executive Summary section summarizes the remedial responses which are likely to be the focus of the FS investigation. Some of the potentially applicable remedial measures for this site are discussed below.

Limited Site Access

One of the immediate measures to be taken at Job Corps may be to limit human and wildlife exposure to the site. Fencing and closing the area to all but Refuge Personnel, and maintaining a thick vegetative

cover, may be appropriate until further remedial action can be commenced. Deed restrictions might be imposed to limit future uses of the area.

Surface Water Control

The purpose of surface water control would be to prevent run-on and run-off within the landfill, to preserve the vegetative cover, and prevent transport of contaminated soil to the pond and eventually to Crab Orchard Lake.

Off-Site Removal or On-Site Containment of Soil and Sediment

Contaminated surface soil and sediments (up to 1-1.5 ft depth) might be excavated and removed for treatment off-site or regraded and contained on-site. Clean soil will be used for fill and capping any areas requiring excavation. Alternatives for containment of wastes might include removal to a secure landfill or secure storage such as in the Munitions Bunkers in Area 13. An estimated 700 CY of soil from the landfill area may require removal or containment based on the sampling results from the RI. The areal extent of contamination at the landfill may extend north and east beyond the area sampled an additional 25-50 ft. on either side; if this is confirmed by pre-remediation sampling, the total volume of soil for removal could be greater than 700 CY.

The sediments from the shallow areas of the pond adjacent to the landfill contained PCBs, cadmium and lead concentrations which exceeded the levels detected at the control sites. These sediments pose a concern in that contaminants can slowly leach or suspend with sediments in the water and increase the probability of exposure to wildlife and humans or

could be transported off-site. Based on the sampling program, the extent of sediment contamination in the pond is estimated to include an area extending 25 ft. from the shoreline surrounding the landfill up to a depth of 2 ft. However, this extension might be better defined prior to actual cleanup. Based on the RI data, an estimated 560-600 CY of sediment may require removal. The sediment samples collected from the deep (middle) portion of the pond did not contain elevated levels of contaminants.

The required depth and areal distribution of contaminated sediments and soil requiring cleanup might be further defined through additional sampling. The required sampling efforts might be incorporated in the field efforts proposed for the FS.

Monitoring

The remedial response alternative to be implemented at this site might include periodic sampling and analyses of the four monitoring wells and of the pond water and sediment for cadmium, lead, and PCBs. Follow-up studies might begin immediately after remediation and continue periodically to verify the adequacy of the cleanup.

24.6 Conclusions and Recommendations

It can be concluded that the Job Corps Landfill is impacted, with the primary pollutants being PCBs and lead. Based on a quantitative risk assessment, it was found that, since human exposure is limited, due to the location of the site, concerns for protection of wildlife would be the focus of the remediation effort.

SECTION 25 - SITE 18, AREA 13 LOADING PLATFORM

25.1 Site Description

Area 13 of the Refuge consists of approximately 85 bunkers that were originally built for storage of 500 lb. bombs. Most of the bunkers are currently used by Olin Corp. and U.S. Powder to store explosives. Agricultural fields are cultivated between the bunkers. This area was served by a rail spur which was abandoned and dismantled. It was reported to the Refuge Manager that chemicals used in munitions manufacturing were dumped off the platform.

Site 18 consists of the Area 13 loading platform, a concrete pad 235 feet long by 10 feet wide elevated by about five feet. (See Figure 25-1). The dock is supported on concrete posts spaced about 9 feet apart. The northwest side of the platform contains stone bedding (probably from the old railroad bed) with a number of small areas of ponded water. No unusual vegetation changes were detected. The only curious item was a pile of dirt and stone rubble off the west end of the dock with a rusted drum shell nearby.

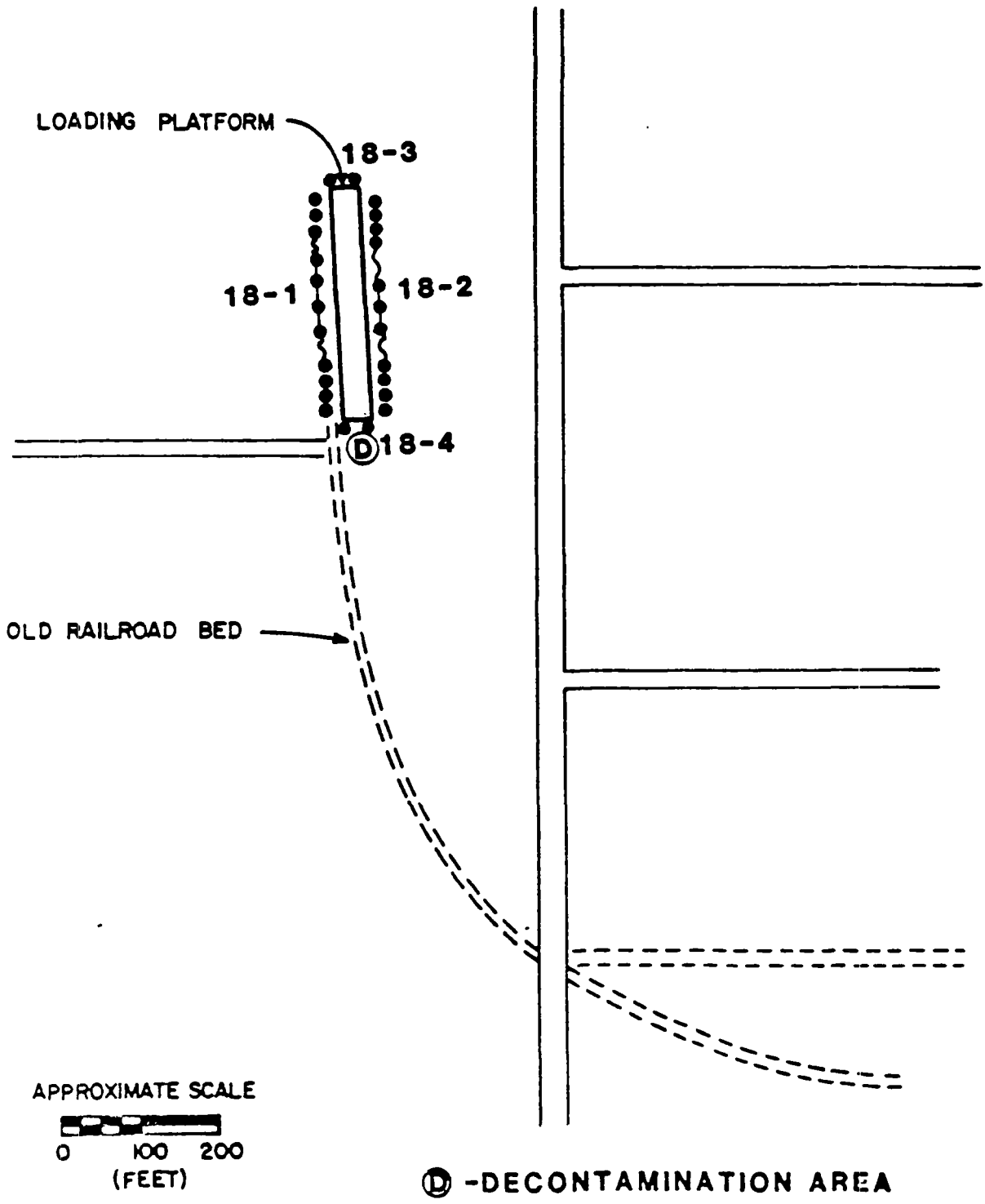
25.2 Site Investigations

25.2.1 Phase I Site Investigations:

Four composite soil samples (0-1 ft depth) were collected around the perimeter of the dock. Samples were collected from each of the two sides of the dock and from each end. The composites along the north and south consisted of 20 grabs, while the east and west composites consisted of 2 grabs each. Location 18-4, at the west end of the dock, was resampled for full priority pollutant analysis because it contained the highest FID scan reading at Site 18.

FIGURE 25-1

SITE 18
AREA 13 LOADING PLATFORM
PHASE I



O'BRIEN & GERE

25.2.2 Phase II Site Investigations:

No sampling was conducted in the Phase II investigation.

25.3 Analytical Results (See Appendix I, Page 18)

Trace quantities of the explosive tetryl were observed in two soil samples (1.90 mg/kg in each) from the north and east sides of the platform. Metals, volatiles and indicator parameter concentrations were similar to concentrations found in soils from the control sites. Two exceptions were noted: magnesium, 91,100 mg/kg in sample 18-4 and sodium, 2,330 mg/kg in sample 18-1, although these are estimated values only. CLP HSL organics analyses on sample 18-4 showed the presence of 4,050 ug/kg wet weight di-n-octyl phthalate. Several other semi-volatile organics were also detected at concentrations less than 300 ug/kg. Acetone and methylene chloride were detected, due to contaminants in the laboratory QA/QC blank. The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values *and some compounds which were not detected may* in fact be present.

25.4 Environmental Effects

25.4.1 Qualitative Assessment

This site was chosen for investigation based on its history of use as a loading dock for chemicals and explosives used in munitions manufacturing. It has also been reported that chemicals have been dumped on the site. The use of nearby areas as agricultural fields also warrants the need for an accurate site characterization.

Phase I analyses detected traces of the explosive tetryl. All other parameters were similar to the concentrations found in soils from the

control sites with the exception of magnesium and sodium in two different samples. However, at these concentrations, these metals will not threaten wildlife or affect human health. Di-n-octyl phthalate was also found in one sample, but it too was at a concentration below the Refuge background level.

Because there is no established waste source at this location, it is not possible to have a "complete" exposure scenario. Therefore, on the basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

25.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

25.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a site inspection and a sample analysis. The only unusual observation was a rusted drum shell located on a pile of dirt and stone rubble, indicating that some supplies may have been dumped when the loading platform was in use.

Chemical residue information consisted of analytical results for surface soil samples. This information was obtained only for the top one foot of soil; deeper soil borings were not conducted. Since there is no evidence to suggest that the surrounding soil has been disturbed, and the loading and unloading activities practiced at the site would likely

contribute only to surface contamination, these samples should adequately represent the site conditions.

It can be concluded that the data generated are adequate for evaluation of the remedial alternatives for this site. The analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

25.5 Preliminary Remedial Alternatives

Preliminary Phase I screening results discussed in the previous section indicated that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore this site was not included in the Phase II investigations. There will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

25.6 Conclusions and Recommendations

It can be concluded that the Area 13 Loading Platform does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

SECTION 26 - SITE 19, AREA 13 BUNKER 1-3

26.1 Site Description

Further information on Area 13 can be found in Section 25.1. It has been reported that chemicals may have been released near Site 19, Bunker No. 1-3, probably in the adjacent field. There is no observable impact on vegetation in the field except for one area of discolored vegetation. Wildspread presence of scattered red bricks suggests dumping has occurred at the site. An L-shaped area of brown vegetation was noted to the west side of one of the bunkers.

26.2 Site Investigations

26.2.1 Phase I Site Investigations:

Four composite soil samples (0-1 ft depth) were collected, one from each side of the bunker at distances up to about 125 feet. (See Figure 26-1). An additional composite soil sample was taken from the brown vegetation area.

26.2.2 Phase II Site Investigations:

One Phase I soil location (0-1 ft depth) was resampled for mercury analysis.

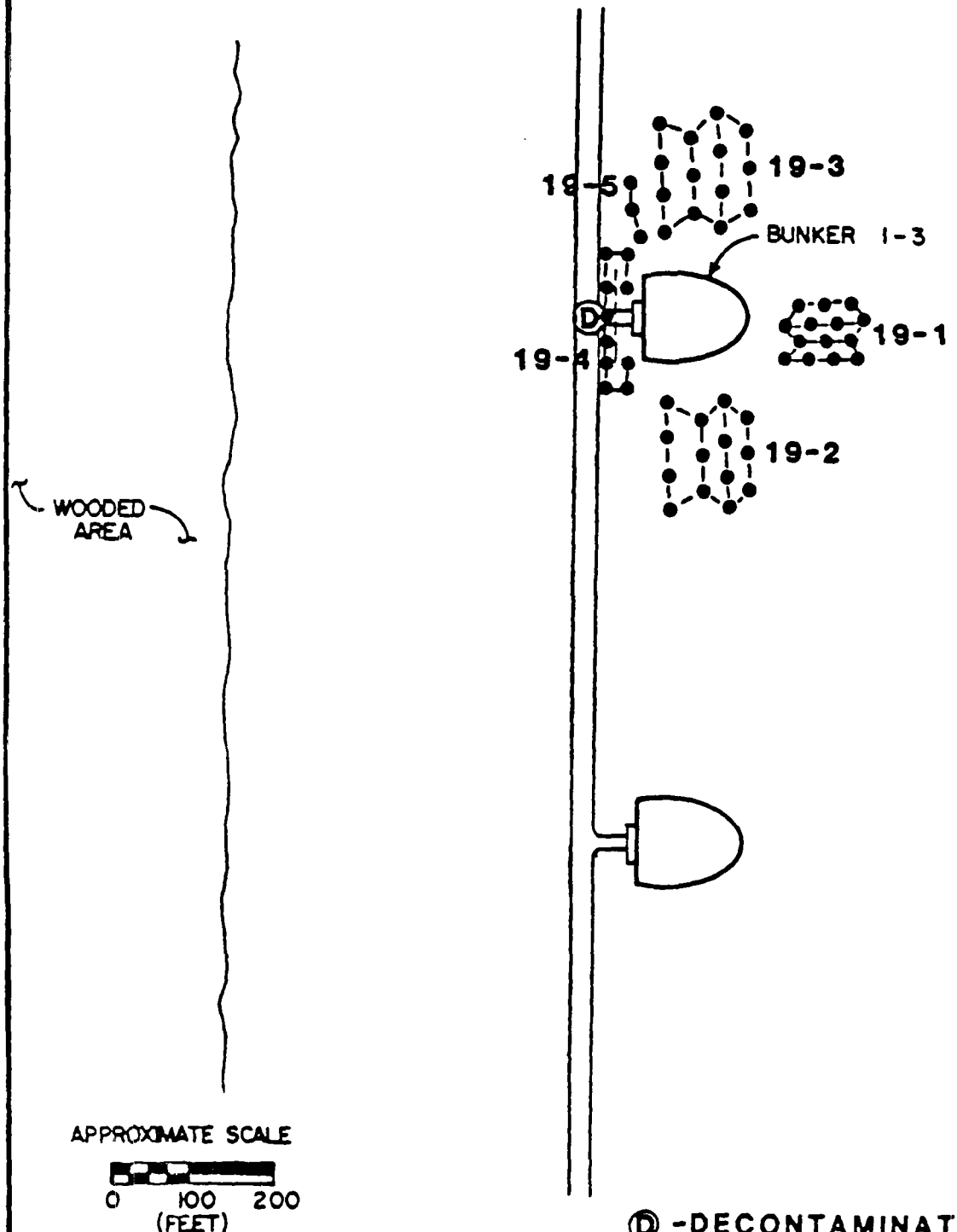
26.3 Analytical Results (See Appendix I, Page 19)

26.3.1 Phase I Analytical Results:

The FID scans showed low organic levels, on the order of 286-1,901 ug/kg, in the soils. The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support

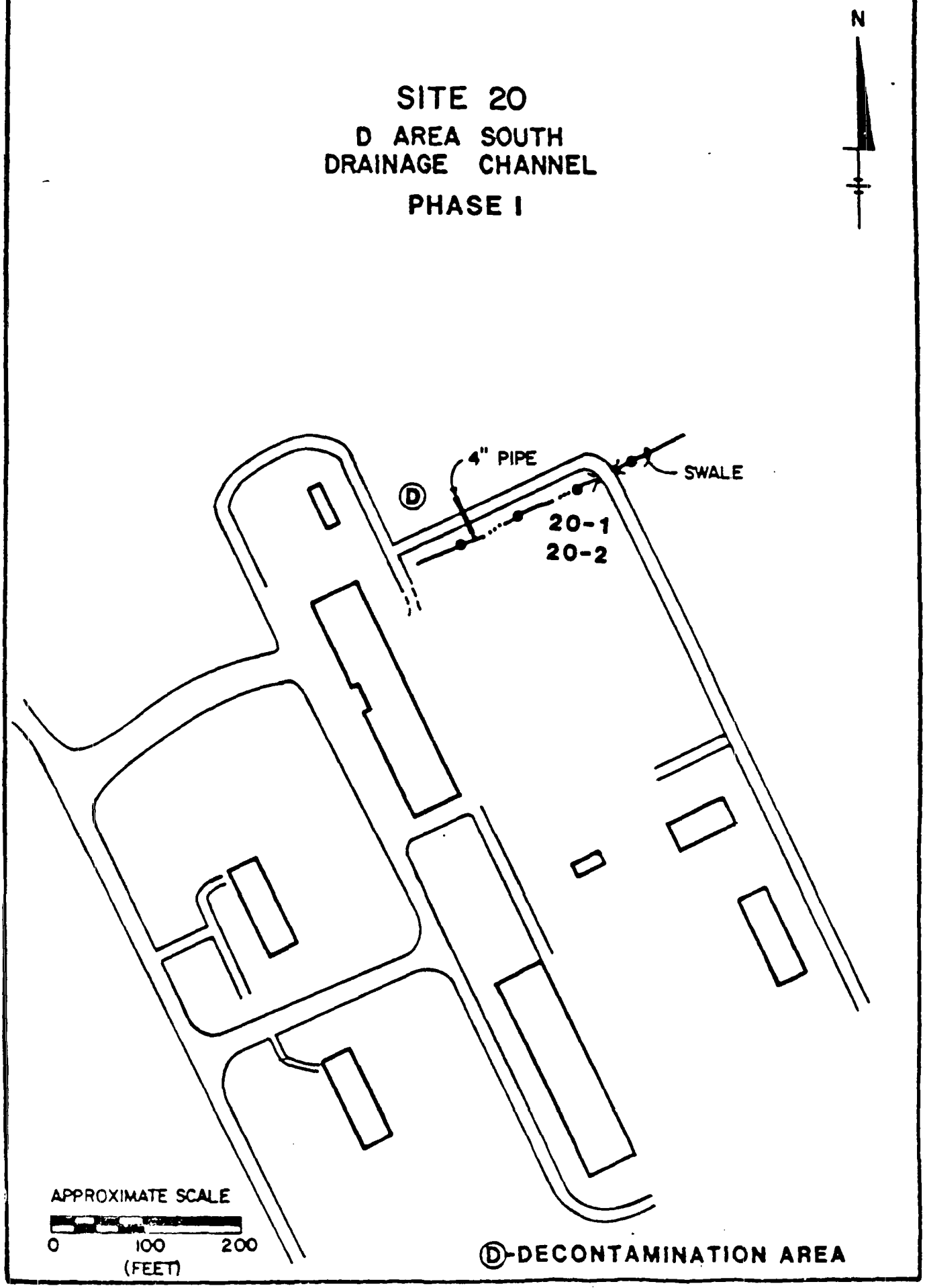
FIGURE 26-1

SITE 19
AREA 13 BUNKER 1-3
PHASE I



O'BRIEN & GERE

FIGURE 27-1



O'BRIEN & GERE ENGINEERS INC

data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present. The sample analyzed for full organics contained, on a wet weight basis, PCBs (1.1 mg/kg) and N-nitrosodimethylamine (1,455 ug/kg). The sample collected from the area of brown vegetation (location 19-5) contained 0.90 mg/kg of the explosive tetryl. Mercury was detected in one sample (19-4), from the front of the bunker, at 3 ug/kg, but this analysis was repeated in Phase II due to poor calibration data. All concentrations for metals in soils were similar to those detected at the control sites.

26.3.2 Phase II Analytical Results:

The mercury concentration in the second sampling of location 19-4 was 28 ug/kg, slightly above the detection limit of 20 ug/kg.

26.4 Environmental Effects

26.4.1 Qualitative Assessment

This site was chosen for investigation based on reports that chemicals had been poured onto the ground in an adjacent field. This dumping may have contaminated the surrounding area, as evidenced by a patch of discolored vegetation and an L-shaped area of brown vegetation.

Phase I analyses detected traces of the explosive tetryl and N-nitrosodimethylamine. N-nitrosodimethylamine has been shown to be carcinogenic in a number tests and is considered a suspect human carcinogen on this basis. The concentration of this compound at this site is higher than detected anywhere else on the Refuge, N-nitrosodimethylamine will be the focus of the risk assessment. The

Phase II mercury concentration was slightly above the detection limit but is within the level typically found in soil matrices and is not considered to pose a threat to wildlife or to human health.

26.4.2 Quantitative Assessment

The preliminary data from the Phase I survey indicate that N-nitrosodimethylamine is the contaminant of primary concern at this site, with 1,455 ug/kg detected in a site soil sample. Due to analytical deficiencies noted in the Phase I result for the single soil analyzed for this compound, and in order to ensure a conservative outcome from the analysis, the assessment below is based on twice the concentration detected, or 2,910 ug/kg. Because this compound has been shown to be carcinogenic to animals, a preliminary quantitative risk assessment will be performed, even though the data available are too limited to place a great deal of confidence in the result. Additional areas of uncertainty in the risk assessment are discussed in Section 26.4.3, Analysis of Uncertainties.

It is assumed that, during an excursion through this site, a human visitor to the site may ingest an average of 100 mg of site soil containing 2,910 ug/kg of N-nitrosodimethylamine as a result of direct contact exposure. This contact would result in an average exposure of 0.291 ug of N-nitrosodimethylamine per visit. Given the remoteness of the site and the absence of daily activities by humans in the area (the general area of the bunker sites is restricted by a locked fence to all but authorized personnel), visits to the site by humans do not occur on a daily basis. As a reasonable upper case exposure estimate, it is assumed that a Refuge employee visits the site once per month over a 30-year period, or 360 exposure days. U.S. EPA (1987) has estimated that a daily intake of

0.0137 ug of this compound each day for lifetime may be associated with an upper bound acceptable carcinogenic risk level of 10^{-5}. Utilizing the estimated intake of 0.291 ug/visit, an exposure of 0.004 ug/day is obtained when normalized over a 70 year lifetime. The estimated incremental cancer risk associated with this exposure level is 2.9×10^{-6}. This estimate, developed under a set of reasonable worst case exposure assumptions, is within the range of 10^{-4} to 10^{-7} risk considered acceptable to regulatory agencies for exposed populations.

The detection of N-nitrosodimethylamine in a soil sample analyzed in the Phase I survey of this site also presents a mechanism for exposure for terrestrial wildlife via the direct contact route. The levels of exposure would be greatest amongst small mammals as a result of inadvertent ingestion of contaminated soil residues and dust during daily burrowing, feeding and grooming. Thus, the risks of direct contact of these species to site nitrosamines residues are assessed. The risks to larger and/or less sensitive species, or to those which have less contact with soil residues would be proportionately lower. A search of on-line data bases (Pollution Abstracts, Biosis Previews, NTIS, HSDB) did not identify published studies on the effects of N-nitrosodimethylamine on pertinent wildlife species. Therefore, tests with surrogate species (i.e. laboratory rodents) are used in the assessment below.

Using the exposure assumptions detailed in Section 24.4.2.2 for exposures of burrowing mice, exposure to soil residues of 2.91 mg/kg N-nitrosodimethylamine will produce a daily intake of 2.76×10^{-2} mg/kg/day by the ingestion and inhalation routes. The most widely reported effect of chronic exposure of laboratory rats and mice to N-nitrosodimethylamine is the induction of hepatocellular carcinoma in a

number of investigations (HSDB, 1987). Using these data, U.S. EPA (1986) derived a unit risk factor of 26 $(mg/kg/day)^{-1}$. This value is an upper bound estimate of the slope of the tumor-exposure relationship, chosen as a conservative estimate of human response to carcinogen exposure at low concentrations. Assuming a similar sensitivity for wild rodent species, a cancer risk estimate of 0.72 is derived for burrowing rodents at this site. The significance of this risk level is uncertain. As discussed by Newell et al.(1987), concerns regarding the effects of cancer on wild populations are largely unknown, and risk levels of concern to humans are not directly transferable to wildlife. Many other factors come into play when addressing whether a wildlife population can maintain itself (i.e. survival to reproductive age, competition, weather, disease, predation, etc), and the effect of cancer, generally forming later in an exposed organism's lifetime, might thus be very small. On this basis, Newell et al. (1987) chose a risk level of 10^{-2} as a level of acceptable carcinogenic risk for wildlife, with the acknowledgement that more study is needed to justify this choice. Using this rationale, it is concluded that wildlife exposure to site residues of N-nitrosodimethylamine could result in a carcinogenic response, but the biological significance of the response cannot be assessed with currently available information.

Additional review of the literature on the effects of N-nitrosodimethylamine to wildlife is presented in USEPA's study on the Environmental Effects Profile on Nitrosamines (1986--). The authors used a reproductive effects study in mice to establish a minimum effective dose for N-nitrosodimethylamine. In this study, female mice were exposed to 0.1 mg/L of N-nitrosodimethylamine in drinking water for 75 days prior to mating, through pregnancy and weaning. Such exposure resulted in

significantly elevated fetal mortality in the treated group. Since mice consume approximately 5.7 ml of water per day, and the adult mouse body weight is approximately 30 g, an approximate exposure rate of 0.019 mg/kg/day is obtained as a lowest observed effect level (LOEL). The estimated daily exposure rate of 0.0276 mg/kg/day using twice the residue level detected at this site is roughly equal to this LOEL. Thus, assuming exposed species are of equivalent sensitivity to N-nitrosodimethylamine as laboratory mouse strains, and that the presence of this compound is widespread through the site at twice the level quantified in the survey, the estimated exposure rates may elicit a toxic response under subchronic conditions.

26.4.3 Analysis of Uncertainties

The information relied upon for evaluating this location consisted of a site inspection and sample analyses. The inspection of the site revealed one area of discolored vegetation and some evidence of dumping. Chemical residue information consisted of analytical results on surface soil samples. This information was obtained only for the top one foot of soil, whereas deeper soil borings were not conducted. Since there is no evidence to suggest that the surrounding soil has been disturbed, and the storage activities at this site would most likely not contribute to subsurface contamination, these samples should adequately represent the conditions of the site.

The quantitative risk assessment was performed under a worst case chronic exposure scenario of repeated monthly exposures and considered only the single data point available for a soil sample in which N-nitrosodimethylamine was detected in the Phase I survey. This result is

only qualitatively reliable due to insufficient QA/QC supporting the analysis. In addition, the residue concentration detected was doubled in the risk calculation to provide a more conservative outcome. This worst case approach resulted in a risk estimate on the lower end of the 10^{-6} to 10^{-4} level generally considered as an acceptable range of risk to humans. The exposure level estimated for inherent populations of small burrowing mammals could measurably affect reproduction of such species if these were to meet the set of worst case assumptions used in the assessment. However, the actual level of risk posed by site-related contaminant exposure cannot be stated with confidence without a more thorough sampling of the site.

26.5 Preliminary Remedial Alternatives

The [illegible] results and the evaluation of environmental effects for this site indicated that N-nitrosodimethylamine residue levels in soil may pose unacceptable exposure levels to small wild rodent species. The risks to potential human receptors were determined to be on the order of 10^{-6}, which is a level generally considered acceptable. Due to previous activities in this area, including use of the bunker sites for storage of munitions, explosives, and other wartime supplies, the nitrosamines residues in a soil sample from this site may be the result of degradation of chemicals previously stored in this area. No further evaluation of remedial alternatives will be conducted for this site; however, it is recommended that the Refuge Management initiate additional investigations in this area to determine if further action will be necessary to protect potentially exposed wildlife.

26.6 Conclusions and Recommendations

It can be concluded that the Area 13 Bunker 1-3 site does not represent a risk of exposure to human health, but may be affecting resident wildlife due to the presence of low level N-nitrosodimethylamine residues in soil. It is recommended that additional investigations be initiated to further evaluate the potential risks associated with residues at this site. This site will not be evaluated further as part of this RI/FS.

SECTION 27 - SITE 20, D AREA SOUTH DRAINAGE CHANNEL

27.1 Site Description

Area D is an active Olin operation located north of Crab Orchard Lake. This area is currently used for the manufacture of explosives. The site was previously used by Universal Match under contract to the DOD. Their operations ceased after a large explosion, according to the Refuge Manager.

An abandoned building is located within the fenced southeastern end of the Olin D Complex. It was reported that chemicals were dumped here. Site 20 consists of a drainage swale originating at the building that runs east outside of the fence. (See Figure 27-1). A four-inch pipe (dripping at the time of the site inspection) extends from the Olin Area under the fence and discharges to this ditch. A slight sheen was noticeable on the surface water in pooled areas of the ditch.

27.2 Site Investigations

27.2.1 Phase I Site Investigations:

One sediment composite of four grab samples (0-1 ft depth) was collected. The sediment was resampled for full organics analysis. One water sample was scheduled but could not be collected because the ditch was dry.

27.2.2 Phase II Site Investigations:

Cyanide and mercury analyses were scheduled for one water sample from the ditch; however, the ditch was dry at the time of sampling and the sample could not be collected.

27.3 Analytical Results (See Appendix I, Page 11)

Cyanide (13 mg/kg) and mercury (8.9 ug/kg) were detected in the sediment composite, however, these data are questionable due to QA/QC deficiencies. The sediment was analyzed for full CLP organics after an FID screen of 16,477 ug/kg; it contained 30,500 ug/kg wet weight di-n-octyl phthalate, 2320 ug/kg wet weight bis (2-ethylhexyl) phthalate, and 336 ug/kg wet weight N-nitrosodimethylamine. All other organics were below the detection level or were reported but were also present in the laboratory QA/QC blank. The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). *The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present.*

27.4 Environmental Effects

Envi[illegible]f drainage within the D and P areas are discussed in Section 16.4.

27.5 Preliminary Remedial Alternatives

Preliminary Remedial Alternatives for the D and P areas are discussed in Section 16.5.

27.6 Conclusions and Recommendations

Conclusions and Recommendations for the D and P areas are discussed in Section 16.6.

SECTION 28 - SITE 21, SOUTHEAST CORNER FIELD

28.1 Site Description

Site 21 is a fenced field (150 ft. x 400 ft.) located at the southeast corner of the Refuge. (See Figure 28-1). The field is thought to be the site of an old dump due to the presence of concrete rubble near one end. No other evidence of debris is observable. The topography gradually slopes to the south and east toward a swampy ditch at the bottom of the slope. Large diameter trees in the field indicate the area has not been disturbed for at least sixty to seventy years.

28.2 Site Investigations

28.2.1 Phase I Site Investigations:

A magnetometer and electromagnetic terrain conductivity survey was conducted along four north-south transects. (See Figures 28-2 and 28-3). Four composite soil samples (0-1 ft depth) were collected, one along each transect. One composite along transect 1 was resampled for full organics analysis.

28.2.2 Phase II Site Investigations:

The transect 1 soil composite was resampled and analyzed for mercury.

FIGURE 28-1

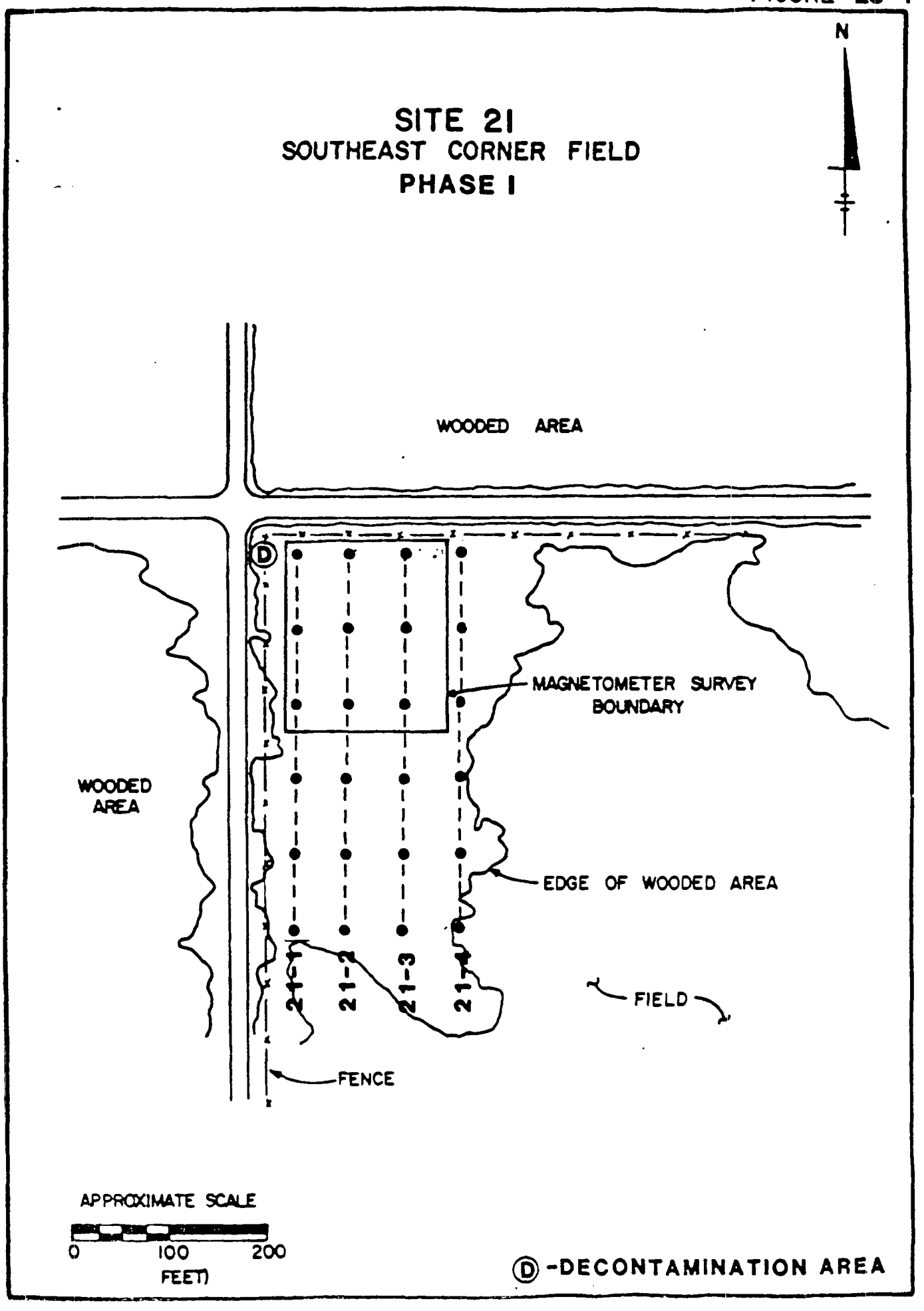
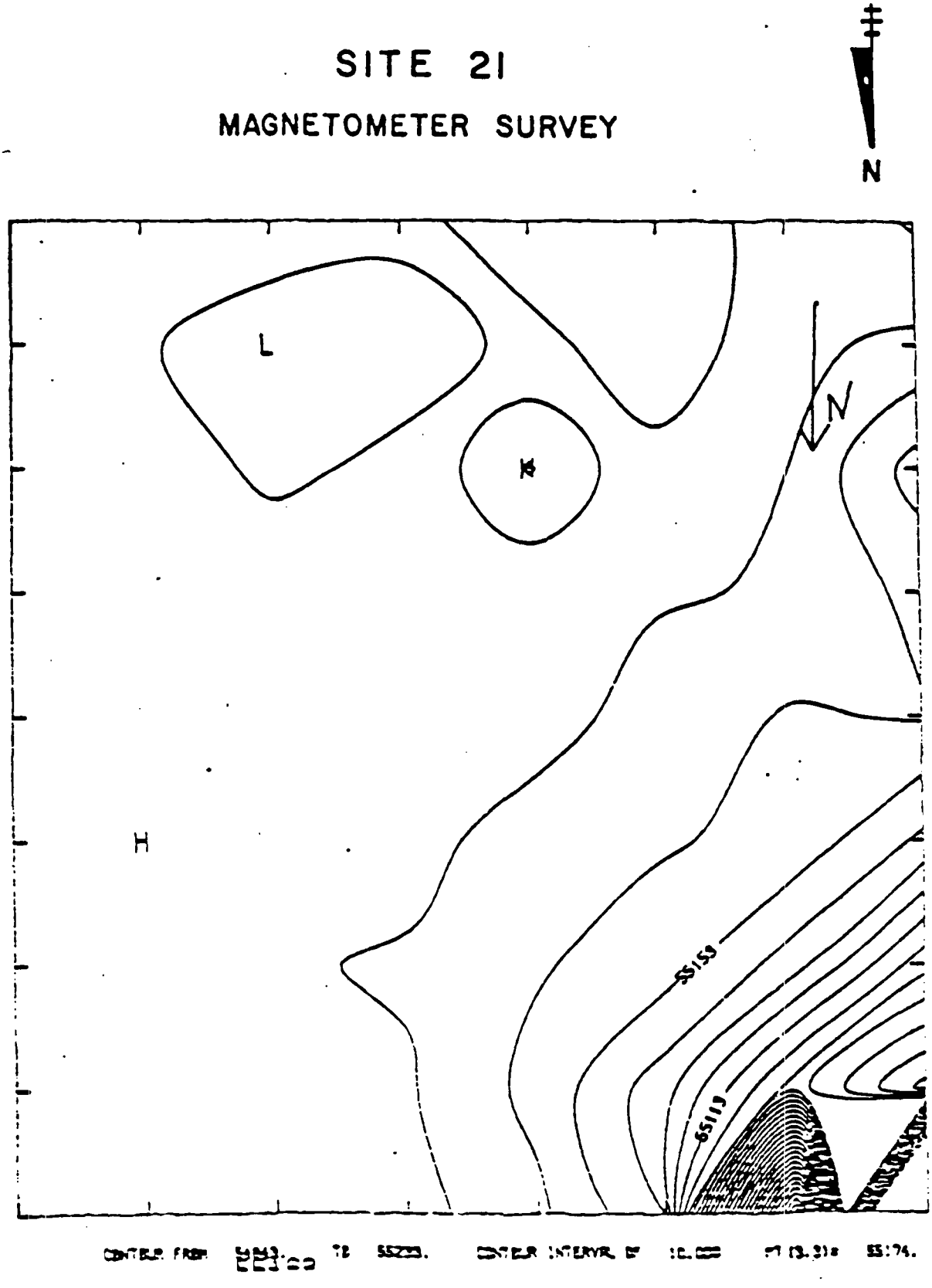

N
SITE 21
SOUTHEAST CORNER FIELD
PHASE I
WOODED AREA
WOODED AREA
MAGNETOMETER SURVEY BOUNDARY
EDGE OF WOODED AREA
FIELD
FENCE
21-1
21-2
21-3
21-4
D
APPROXIMATE SCALE
0
100
200
FEET)
D -DECONTAMINATION AREA

FIGURE 28-2

SITE 21

MAGNETOMETER SURVEY

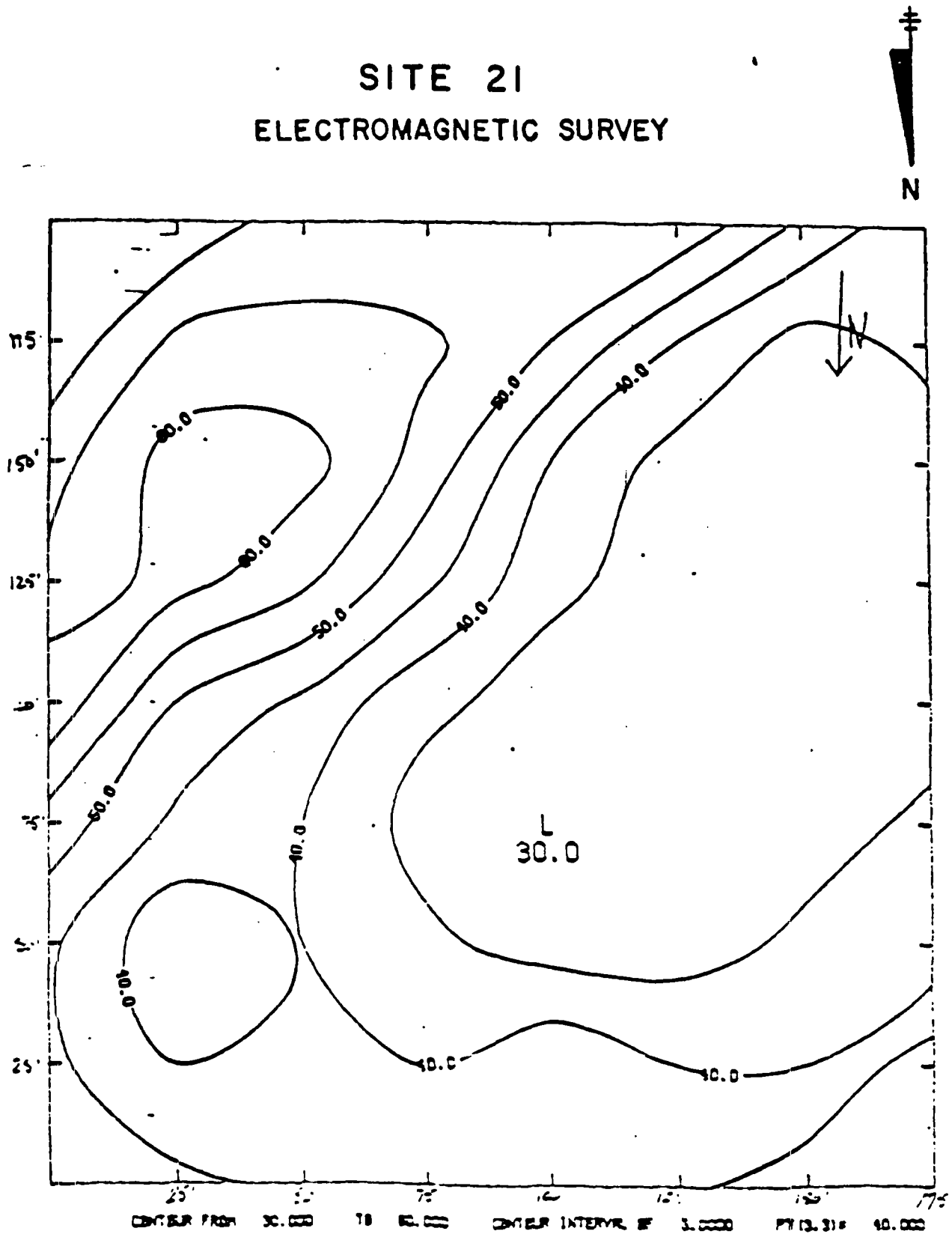


CONTOUR FROM [illegible] TO 55223. CONTOUR INTERVAL OF 10.000 AT (3.3)0 55174.

FIGURE 28-3

SITE 21

ELECTROMAGNETIC SURVEY



O'BRIEN & GERE

28.3 Analytical Results (See Appendix I, Page 22)

28.3.1 Phase I Analytical Results:

The magnetometer and electromagnetic survey as shown in Figures 28-2 and 28-3 did not indicate any unusual subsurface conditions.

Two of the soil samples contained high magnesium levels (10,500 and 27,200 mg/kg, wet weight), which were approximately one order of magnitude higher than the levels detected at the control sites (metals are estimated values only). The FID scan on the composite soil sample from transect 1 was 20,630 ug/kg (25,274 ug/kg duplicate); however, only trace base/neutral extractable compounds were detected. The semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present. N-nitrosodiphenylamine (156 ug/kg wet weight) was the only compound detected above the detection limit, although traces of other semi-volatiles were also reported. The sample representing transect 1 also contained 9 ug/kg of mercury; all other soil samples contained mercury below the 1 ug/kg detection limit. The mercury analysis was repeated in Phase II due to poor calibration data.

28.3.2 Phase II Analytical Results:

The soil composite contained 41 ug/kg of mercury.

28.4 Environmental Effects

28.4.1 Qualitative Assessment

This site was chosen for investigation based on the thought that it had been an old landfill at one time. The site also slopes towards a swampy drainage ditch, which would be a viable transport mechanism, should the site be contaminated.

Phase I sampling analysis detected traces of magnesium, but the concentration was below the detection limit. N-nitrosodiphenylamine was detected but at concentrations over ten times below those detected at other sites where this compound was not considered to represent a significant risk of exposure (see Section 19.4). One sample also contained traces of mercury and a Phase II soil sample was taken to more accurately quantify this result. The Phase II analysis showed a slightly higher concentration than the Phase I result, but is not considered to be detrimental to the environment.

Because there is no established waste source at this location, it is not possible to have a "complete" exposure scenario. Therefore, on the basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

28.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

28.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a site inspection, geophysical surveys, and sample analyses. An inspection of the site led to the belief that the site had been an old dump. However, the geophysical surveys did not reveal any unusual subsurface conditions. Large diameter trees in the field indicated that the area had not been disturbed for at least sixty to seventy years.

Chemical residue information consisted of analytical results on surface soil samples. This information was obtained only for the top one foot of soil; deeper soil borings were not conducted. Based on the magnetometer and electromagnetic terrain conductivity surveys, there is no evidence to suggest that waste had been buried on this site. Therefore, these samples should adequately represent the conditions of the site.

It can be concluded that the data generated are adequate for evaluation of the remedial alternatives for this site. The sampling analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

28.5 Preliminary Remedial Alternatives

The analytical results discussed in the previous section indicate that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore there will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

28.6 Conclusions and Recommendations

It can be concluded that the Southeast Corner Field site does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

SECTION 29 - SITE 22, OLD REFUGE SHOP

29.1 Site Description

North of the Refuge along Wolf Creek Road is the old Refuge Headquarters, now leased by Diagraph-Bradley. Site 22, the Refuge Shop, was located behind the Headquarters building. Pine wood poles were treated in a fenced area of the Shop with pentachlorophenol wood preservative and shipped to various locations throughout the county, according to the Refuge Manager. A small drainage pool is located outside the fence to the north and contains a green-yellow scum. (See Figure 29-1). The pool drains through the woods to the northwest and ultimately into Crab Orchard Lake.

29.2 Site Investigations:

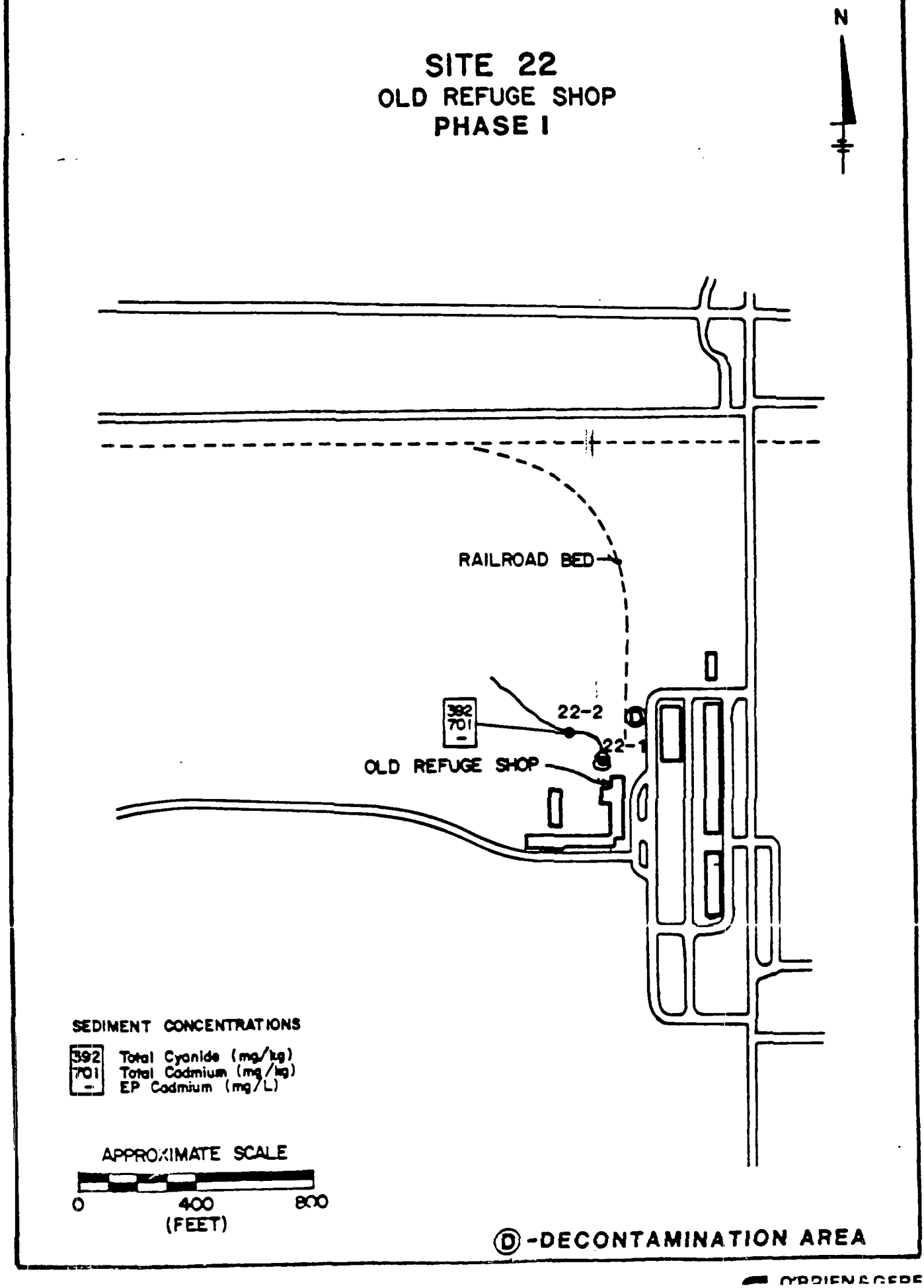
29.2.1 Phase I Site Investigations:

One grab surface water sampled from the drainage pool. One composite sediment sample (0-1 ft. depth) was collected from the drainage ditch. The sediment was resampled for full priority pollutant analyses.

29.2.2 Phase II Site Investigations:

A monitoring well was installed and sampled during Phase II. The monitoring well was set to a total depth of 10 feet in silty clay and was screened from 5 to 10 feet. The ground water was sampled and analyzed for CLP HSL volatiles, base/neutral/ acid extractables, and metals.

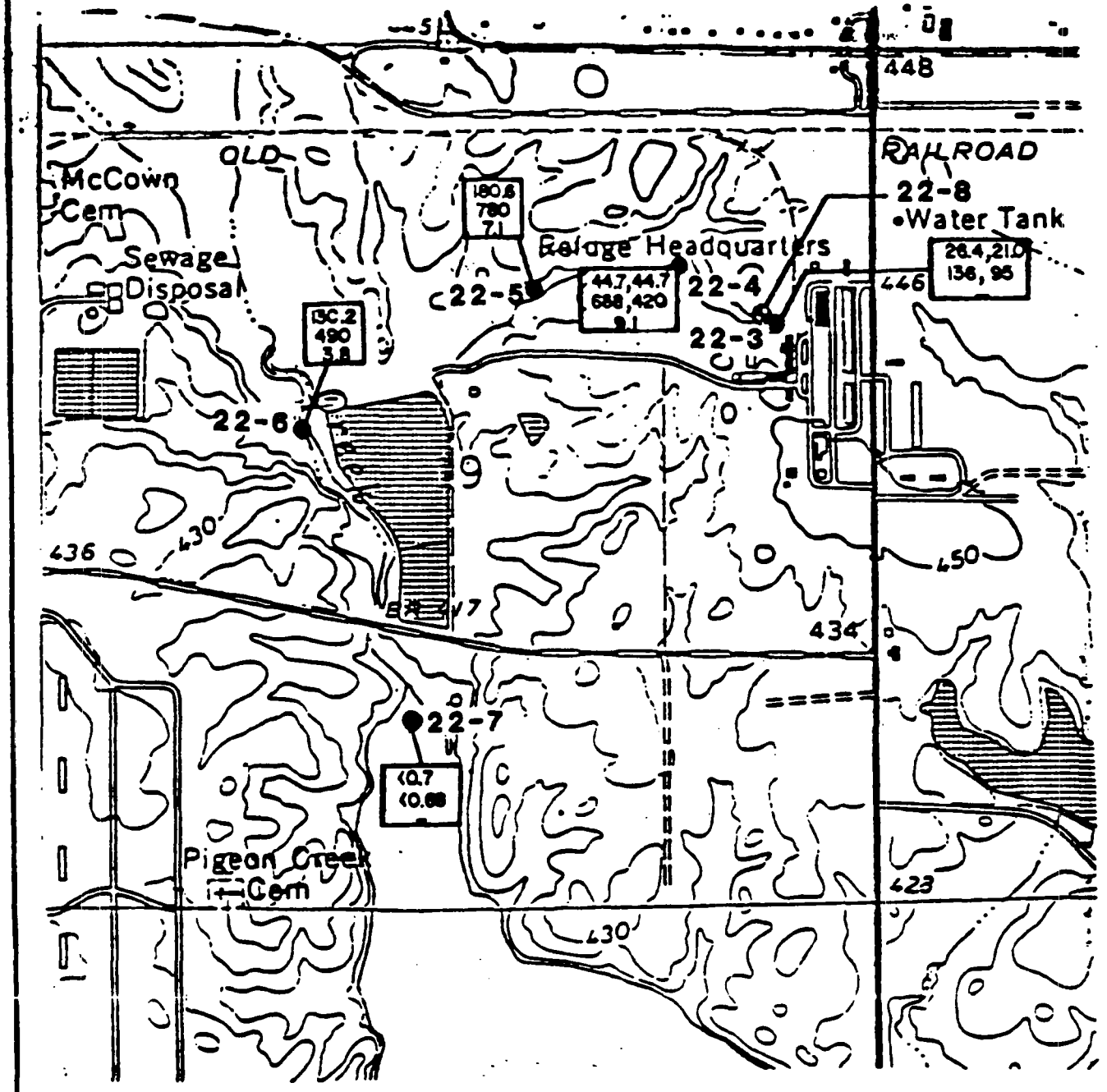
FIGURE 29-1



O'BRIEN & GERE

FIGURE 29-2

SITE 22
SAMPLING LOCATIONS
PHASE II



Four sediment samples were collected from the ditch. One additional soil sample was collected from the embankment to trace the downstream distribution of contaminants (see Figure 29-2). The soil and sediment samples were analyzed for CLP base/neutral/acid extractables, as well as cadmium, chromium, and cyanide. EP-toxicity analyses were completed on three of the sediment samples.

29.2.3 Site Hydrogeologic Characterization

29.2.3.1 Site Geology

Based on results of the test well boring 22-8, the subsurface unconsolidated overburden consists of a brown, gray, and orange mottled silty clay, with some sand. This material is present from the ground surface to 9 ft. in depth. Beneath the silty clay, at least 1 ft of a brown silt with a trace of fine gravel is present to 10 ft in depth (total depth of boring). Bedrock was not encountered in the boring; therefore, the depth to bedrock and bedrock lithology is unknown. As only the one monitoring well was installed, the lateral extent and variability of the overburden units is also unknown.

29.2.3.2 Site Hydrogeology

Shallow ground water occurring beneath the site was found at a depth of 1 to 1.3 ft. below the ground surface within the silty clay soil unit during June 1987. The monitoring well installed screened this upper water table. Ground water elevations collected during the winter and summer of 1987 (wet and dry seasons, respectively) indicate a water table fluctuation of 0.3 ft with water levels dropping during the summer

months (Table 4-3). Figure 35-5 illustrates the monitoring well location and the ground water elevation of 18 June, 1987.

29.3 Analytical Results (See Appendix I, Page 22)

29.3.1 Phase I Analytical Data:

The pool water sample did not contain contaminant concentrations above the Illinois Public Water Supply Standards or Federal drinking water standards. The total organic halides concentration in water was 14 ug/L (16 ug/L duplicate). The sediment contained cyanide (392 mg/kg), cadmium (701 mg/kg), chromium (663 mg/kg), and lead (150 mg/kg) above the concentrations detected at the control sites. The metals concentrations are reported as estimated values and cyanide analyses were repeated due to QA/QC deficiencies (see Exhibit B). All organics were below detection limits, although the FID scan was 10,114 ug/kg. Methylene chloride and acetone were quantified, but these were also detected in the laboratory blank. The total organic carbon concentration in the sediment was 19,413 mg/kg, and organic nitrogen was 1,899 mg/kg.

29.3.2 Phase II Analytical Data:

The ground water sample contained low levels of cadmium (25 ug/L), chromium (21 ug/L), and lead (6.6 ug/L). The corresponding dissolved metals concentrations were 17, 5.1, and 3.9 ug/L. Cyanide was detected at 0.07 mg/L. No organics were detected in the HSL/CLP analysis, although the spike/spike duplicate recoveries were outside of control limits and the semi-volatiles were extracted outside of the holding time. All parameters were within the Illinois Public Water Supply

Standards, except for cadmium and cyanide. The cadmium concentration also exceeded the Federal drinking water MCL and MCLG.

Sediment samples 22-5 and 22-6 downstream contained the highest cyanide concentrations, 130 mg/kg and 181 mg/kg wet weight. The sediments also contained cadmium (less than 0.68 - 780 mg/kg), chromium (10-889 mg/kg) and lead (93 - 166 mg/kg). Chromium duplicate analyses were not within the control limits. In general, the sediments which contained high cyanide levels also contained high metals concentrations. Three sediment samples, extending approximately 3000 ft. downstream of the pool, contained EP Toxic cadmium concentrations (9.1, 7.1, and 3.8 mg/L) in excess of the RCRA criterion of 1.0 mg/L, thus defining these sediments as hazardous wastes. Figure 29-2 shows the cyanide and total and extractable cadmium concentrations in the sediments. Trace base/neutral/acid extractable compounds were found in soil 22-3 (and duplicate 40-76), including 2-methylnapthlalene (330 and 280 ug/kg), bis(2-ethylhexyl) phthalate (200 and 320 ug/kg), and di-n-butyl phthalate (1,260 and 499 ug/kg), although the recoveries for spike and spike duplicate samples were outside of limits for the semi-volatiles scan. Other detected organics were also present in the QA/QC blanks.

29.4 Environmental Effects

29.4.1 Qualitative Assessment

29.4.1.1 Source Evaluation

The results of the site investigations, as described in the preceding sections, determined that the vicinity of the Old Refuge Shop (previously used to treat wood poles with preservative) was contaminated

with residues of cadmium, cyanide, and other compounds. Phase I analyses of water and sediments in a small drainage ditch adjacent to the shop showed the presence of 701 mg/kg cadmium, 603 mg/kg chromium, and 392 mg/kg total cyanide in ditch sediments while the water contaminant levels were less than the Illinois Water Supply Standards. A shallow well installed on the site showed only very low levels of contaminants, but cadmium and cyanide levels exceeded the State and Federal standards. Phase II soil and sediment EP Toxicity analyses at the site and 3000 feet downstream confirmed the site contamination and offsite transport of cadmium. Based on this analytical survey, cadmium and cyanide were chosen to serve as site indicator contaminants for the purpose of this risk assessment.

The physicochemical and toxicological properties of cadmium and cyanide are summarized in Exhibit A. Cadmium is a highly toxic element capable of producing a broad range of systemic effects, particularly to the respiratory, renal, and reproductive systems. It is also a teratogen in animal studies, and there is evidence for carcinogenicity in humans via inhalation as well. Cadmium can accumulate extensively in exposed individuals and in populations through food chain magnification of residues. The major concern from cyanide exposure is acute toxicity of hydrocyanic gas (HCN) and simple salts such as sodium cyanide. HCN would exist as a gas under normal environmental conditions, while the chemistry of the salts is quite complex and compound-specific, with a variety of soluble and insoluble complexes that can be formed. Cyanide does not bioaccumulate.

29.4.1.2 Transport Route Evaluation

a) Air: Cadmium has no propensity to volatilize to air from the adsorbed state, Therefore, transport of cadmium compounds in vapor form is not a significant route of transport. However, because of the existence of exposed soil-adsorbed contaminants, dusts generated by wind erosion, vehicular traffic, or the activities of endemic wildlife constitute a functional route for conveying cadmium residues to on- and off-site locations for subsequent exposures by receptors in those areas. Due to the high vapor pressure of HCN and the relatively low volatility of ionic cyanide, exposures via both the vapor and dust-bound forms of cyanide are possible.

b) Direct Contact: Due to the presence of site indicator contaminants in soils and sediments in the area, exposures by the direct contact route are possible.

c) Surface Water: Phase II analyses detected cadmium residues in excess of 1 mg/kg in sediments 3000 feet downstream of the site drainage pool. Therefore, the surface water transport route is considered functional via precipitation-initiated runoff events which convey soil- and sediment-bound site contaminants towards Crab Orchard Lake.

d) Ground Water: Only minimal quantities of site contaminants were detected in site ground water, and cadmium mobility in silty clay soils is generally low. Furthermore, no exposed receptors (ground water users) were identified for this route. Therefore, the ground water transport route was determined to be non-functional at the site and will not be considered further in this risk assessment.

29.4.1.3 Receptor Evaluation

Human

The Old Refuge Shop is situated in a non-populated area. Therefore, the only potential human receptors would include facility employees, site trespassers, and occasional recreational users of the Refuge. The number of human receptors is low and exposure will be of a transient, non-chronic nature.

Specific scenarios for human exposure to site indicator contaminants will be developed in the following sections. The transport route evaluation identified three functional transport mechanisms: the air route, the direct contact route, and the surface water transport route. Exposures will generally occur only in the vicinity of the Shop and drainage ditch, with the exception of downgradient drainage conveying surface water towards the lake, and potentially from consumption of fish taken from the lake.

The following are the most likely human exposure scenarios for the functional transport routes.

a) Direct Contact: The most probable human exposure scenario would be exposure to site indicators via direct contact with surface residues and sediments at the site and drainage ditch and to sediments in the downstream drainage areas. Humans employed at the facility constitute one group of potential receptors. Occasional recreational users of the Refuge might also be exposed to contaminants via direct contact. The most likely mode of entry of contaminants into the body would be incidental ingestion of soil-bound residues adhering to the skin, clothing, or shoes acquired by direct contact with exposed wastes.

b) Air Route: Dusts generated by wind erosion or foot traffic over exposed waste areas constitute the most likely mechanism for exposure via the air route since cadmium and some forms of cyanide are soil-bound and non-volatile. HCN is highly volatile and thus poses a potential additional exposure mechanism if present at the site. As with the direct contact pathway, the receptors include facility employees and other trespassers who may breathe contaminated dusts while traversing the site.

c) Surface Water Route: As presented in the preceding section, transport of cadmium residues towards Crab Orchard Lake presents a potentially complete human exposure pathway via ingestion of residues accumulated in fish.

d) Ground Water Route: No human users of site ground water were identified. Therefore, this exposure pathway is incomplete.

Wildlife

The forested nature of the site and adjoining areas indicates that a wide variety of terrestrial organisms may be exposed to site-related contaminants, and the proximity to Crab Orchard Lake creates the opportunity for exposures to aquatic populations.

a) Direct Contact: Wildlife inhabiting the site such as invertebrates and small burrowing rodents will receive both acute and chronic direct contact exposures to site contaminants bound to soil dusts during burrowing activities. Exposed waste areas present a functional direct contact route exposure path for birds while feeding, ingesting grit, and dusting. Subsequent ingestion of soil-bound residues while preening or grooming is the principal means of entry into the body.

b) Air Route: Inhalation exposures of wildlife to dust-bound cadmium and cyanide will follow the direct contact scenario described above. In addition, inhalation of HCN vapor is possible.

c) Surface Water: Wildlife using the Shop drainage pool for drinking water will inadvertently ingest sediments containing residues of cadmium, chromium, and cyanide. In addition, transport of cadmium to Crab Orchard Lake via runoff of sediments is a potentially functional chronic exposure pathway for aquatic organisms in Crab Orchard Lake. Exposures will be relatively greatest for benthic invertebrates and bottom-feeding fishes such as catfish. It is noted, however, that cyanide and cadmium were not detected in the sediments of Crab Orchard Lake near the mouth of the stream leading from the Old Refuge Shop. These compounds were, likewise, not detected elsewhere in the sediments of Crab Orchard Lake.

d) Ingestion: Implied in all three wildlife exposure pathways discussed above is the ingestion of site contaminants via soils, dusts, sediments, vegetation, water, and consumed prey. In addition, herbivores may consume contaminated dusts on seeds and vegetation. Fish, birds (i.e. ducks, herons) and other aquatic organisms may inadvertently ingest contaminant-bearing sediments while feeding. The ability of cadmium to accumulate in aquatic and terrestrial food chains is well documented, adding to the importance of the ingestion route of exposure.

e) Ground Water: No surface ground water discharges have been shown which would provide a complete pathway for wildlife exposures.

29.4.2 Quantitative Assessment

29.4.2.1 Estimates of Release and Exposure Rates

Estimates of Exposures by Direct Contact

The qualitative assessment for the Old Refuge Shop has determined that direct contact represents a functional exposure pathway for humans and wildlife. However, cadmium and some forms of cyanide are tightly bound to soils and sediments. Therefore, dermal absorption of contaminants is not expected. The pathway consists, instead, of ingestion of bound residues picked up through direct contact with soils and sediments. The contribution of this route of exposure will therefore be addressed in the section below on ingestion exposures.

Estimates of Airborne Exposures

The qualitative portion of this assessment has established that the air pathway represents a complete exposure route. The pathway consists of breathing contaminated dusts at the site by occasional human activities (visits by employees, recreational users etc.), and by burrowing and dusting activities of wildlife.

The general approach and assumptions used to estimate airborne human and wildlife exposures are given in Section 24.4.2.1 of this report. Using this worst case approach for a four hour excursion by a facility employee or hiker in a sector of the site containing exposed wastes, and assuming a mean surface cadmium soil/sediment level of 500 mg/kg, a total exposure to cadmium of 0.026 mg or 0.37 ug/kg for a 70 kg adult is obtained via the inhalation route per site visit. Assuming three such visits to the site per year, a chronic inhalation rate of 0.003 ug/kg/day is derived. It should be realized that such a scenario does not technically define a chronic exposure. Using the same approach for

exposure to dust-bound cyanide at 100 mg/kg, an exposure of 0.074 ug/kg body weight per site visit is obtained. For repeated exposures, e.g. 3 times yearly, this intake would be expressed as 0.0006 ug/kg/day. The contribution of inhaled residues to total chronic intake is discussed in the following Section 29.4.2.2, Quantitative Assessment.

True chronic inhalation exposures are likely, however, for small burrowing mammals at the site. Assuming an average of 1 hr daily burrowing and breathing using a breathing rate value of 0.006 m3/hour for an active 30 g mouse (approximately U.S. EPA, 1985), four times the resting rate cited in small rodents might receive exposures up to 1.0 mg/kg/day as a result of burrowing in soils containing 500 mg/kg cadmium. Vaporized residues would not be significant due to the very low volability of the cadmium compounds. For 100 mg/kg cyanide, a daily exposure of 0.2 ug/kg/day is estimated. Due to lack of monitoring data, exposure to HCN gas cannot be estimated. Given the acute lethality of this substance, its presence in substantial amounts would be obvious. The significance of this exposure is discussed below in 29.4.2.2.

Additional wildlife species are considered in the following section on Quantitative Assessment.

Estimates of Ground Water Exposures

It has previously been determined that the groundwater exposure pathway is incomplete at the site and therefore will not be considered quantitatively.

Estimates of Surface Water Exposures

In view of a functional transport mechanism for conveying site contaminants towards Crab Orchard Lake via runoff events, the surface water pathway is complete. Exposures due to direct contact with sediments is discussed below under ingestion exposures. Another mechanism of exposure may consist of ingestion of contaminants accumulated in biota from residues present in sediments transported to Crab Orchard Lake. Therefore, exposures by this route will be discussed in the following section on ingestion exposure.

Estimates of Ingestion Exposures

Ingestion exposure of site contaminants at the Refuge Shop and contiguous sites has two components: ingestion of soil-bound residues acquired by direct contact with waste materials, and bioconcentration and foodchain accumulation of cadmium in terrestrial communities and possibly in Crab Orchard Lake. The approaches and assumptions used to estimate exposures by direct contact and ingestion of contaminated soils have been discussed in section 24.4.2, the quantitative assessment for the Job Corps site. Using the worst case assumption that an individual ingests 100 mg of soil as a result of an excursion into an exposed waste area of the site and that the mean surface level of cadmium and cyanide are 500 and 100 mg/kg, respectively, an ingestion of 0.71 ug/kg for cadmium and 0.14 Ug/kg for cyanide are estimated per site visit for a 70 kg human. The corresponding chronic exposure levels of 0.0058 and 0.0011 ug/kg/day could be derived if it is assumed that site visits by humans recur at least 3 times per year.

The following wildlife cadmium and cyanide intake rates from ingestion of contaminated soil at the site during feeding or grooming are estimated using similar assumptions as those detailed in Section 24.4.2 The estimated exposure from 500 mg/kg cadmium in soil are: rabbit, 7.1 mg/kg/day; mouse, 4.75 mg/kg/day; and deer, 2.69 mg/kg/day. For cyanide at 100 mg/kg soil, the corresponding estimates are: rabbit, 1.4 mg/kg/day; mouse, 0.95 mg/kg/day; and deer, 0.54 mg/kg/day.

Beyer et al (1982) determined that earthworms of the family Lumbricidae were capable of bioaccumulating cadmium directly from soil. For example, earthworms living in soil containing 2 mg/kg of cadmium contained levels of cadmium as high as 100 mg/kg body weight. The authors considered these cadmium levels to be hazardous to wildlife which might feed on the worms, although no experimental evidence for this quantitative conclusion was provided. Given the scarcity of data on the effects of ingested cadmium to carnivorous wildlife which might receive such exposures (i.e. shrews, moles, skunks, certain birds, for example), a comparison of possible exposure rates to established effect levels is made in Section 29.4.2.2, Quantitative Risk Assessment for wildlife.

29.4.2.2 Quantitative Risk Assessment

Human Risks

Human exposure at the Refuge Shop site will be limited, with little opportunity for true chronic exposures. Nevertheless, the assessment below carries the scenario through to potential chronic exposures to determine the level of risk to humans. Human exposure estimates for airborne dust-bound cadmium residues (0.003 ug/kg/day) and direct contact ingestion of soil-bound residues of cadmium (0.0058 ug/kg/day)

provide a total estimate of 0.009 ug/kg/day for this scenario. A unit risk factor of 7.8 $(mg/kg/day)^{-1}$ has been established by U.S. EPA (Exhibit A) for assessing human carcinogenicity based on evidence that inhaled cadmium has produced respiratory cancer in the workplace. Using this value and the estimated inhalation exposure rate of 0.003 ug/kg/day, an incremental risk of 2.3×10^{-5} is estimated. This upper limit estimate is very close to the 10^{-5} to 10^{-6} population risk considered by regulatory agencies to constitute a negligible risk to the national population. In view of the low probability that even one receptor would meet all the upper bound assumptions used, the human risk at this site attributable to cadmium exposure is deemed negligible.

Assuming a mean soil/sediment cyanide level of 100 mg/kg and an exposure scenario as just described, a total human ingestion and inhalation exposure rate of 0.214 ug/kg/visit is obtained. An acceptable daily (chronic) intake of 108 ug/kg/day has been established for human cyanide intake (USEPA, 1980); therefore, the worst case acute exposure for one visit to this site is over 500 fold lower than the level which might begin to present concerns for toxicity.

An additional avenue for human exposures is consumption of fish taken from Crab Orchard Lake. Cadmium-contaminated sediments provide a theoretical source for bioaccumulation of residues. This exposure is not quantifiable due to lack of data on fish residues and proof that residues have indeed been transported to the lake.

Wildlife Risks

Estimates of total cadmium and cyanide intakes estimated in the previous sections for receptor species of wildlife are summarized below:

Estimated Daily (Chronic) Intake - Cadmium

	Body Weight (kg)	Inhalation Rate ug/kg/day	Ingestion ug/kg/day	Total mg/kg/day
Deer	60	0.43	2.69	2.69
Rabbit	1.0	0.415	7.1	7.1
Mouse	0.03	1.0	4.75	4.75

Estimated Daily (Chronic) Intake - Cyanide

	Body Weight (kg)	Inhalation Rate ug/kg/day	Ingestion ug/kg/day	Total mg/kg/day
Deer	60	0.087	0.54	0.54
Rabbit	1.0	0.083	1.4	1.4
Mouse	0.03	0.2	0.95	0.95

NOTES: See Table 24-1 and Section 24.4.2.2 for assumptions.
Inhalation rates based on estimated breathing rates for each species, 10 mg dust inhaled per m3 of air, 0.5 ug mean Cd or 0.1 ug mean CN per mg dust, and 1 hr (burrowing animals) or 4 hrs (deer) exposure duration each contact.

These estimates indicate that wildlife exposures to site-related residues may be relatively greatest among small mammals on the landfill such as rabbits, mice, chipmunks, and the like. The latter species animals will be exposed primarily via ingestion of contaminated soil and dust while burrowing, grooming, and feeding on dust-bearing seeds and invertebrates. Rabbits and other herbivores at the site receive the major part of their exposure from contaminated vegetation. Given the broad range of demonstrated possible toxic effects, the potential for interspecific sensitivity, and limited data on effects of cadmium and cyanide on wildlife species, it is difficult to gauge the significance of these exposures. Using data from controlled tests with laboratory

animals, levels of cadmium and cyanide at this site may be sufficiently high to present concerns for reproductive effects and other systemic toxicity in vertebrate species.

U.S. EPA (1986) reports a chronic acceptable intake level for ingested cadmium to be 2.9×10^{-4} mg/kg/day, based on the lowest level which has produced kidney toxicity in humans and incorporating a ten-fold margin of safety. A small (30 g) animal consuming 5 percent of its body weight daily in the form of earthworms or comparable invertebrates might thus eat 1.5 g of food per day. If the worms contained cadmium at 100 mg/kg from living in soils with 2 mg/kg, a daily cadmium ingestion rate of 5.0 mg/kg/day can be calculated. This exposure rate is well in excess of a possible effect level. Since some sediments in the Old Refuge Shop study area contain cadmium of up to 780 mg/kg, exposure to exposed sediments combined with food chain cadmium accumulation could impact localized wildlife populations. Areas of high sediment cadmium are comparatively small, which would serve to lessen the probability for lifetime chronic exposures and overall impact of population exposures, however.

29.4.3 Analysis of Uncertainties

A principal area of uncertainty exists in addressing the risks posed to wildlife by chronic exposure to contaminants at the site. A lack of documentation on the effects of site contaminants on wildlife species which might be found on the site necessitated the use of studies involving laboratory rodents and rabbits as surrogates. The relative sensitivity of these species is unknown.

Regarding the estimate of increased cancer risk in humans from inhaled cadmium, it should be noted that the chronic lifetime exposures assumed are highly unlikely and any exposed population, chronic or otherwise, is very small. In this light, the worst case increase in human risk is considered negligible.

29.5 Preliminary Remedial Alternatives

The environmental effects discussion in Section 29.4 support the need for remedial action for the Old Refuge Shop. The contaminants identified as probable concerns were cadmium, chromium, and cyanide. Lead was detected in soil but at levels within the range for Refuge background. The contaminants were present mainly in the ditch sediment samples, although cadmium and cyanide were detected in the ground water samples and were found at concentrations above the Illinois Public Water Supply standards and Federal drinking water standards. Contamination in sediments was detected in the surface samples (approximately to 1 ft. depth), but no core samples were collected to quantify the levels in deeper sediments.

The risk-based evaluation in Section 29.4 suggests that the risk levels to wildlife species from cadmium and cyanide residues could be greater than the risks to humans, although there are limited data on the toxicological effects to the variety of wildlife species that could be exposed at the Refuge. The worst case assumptions used for the assessment of human risks determined that cadmium levels in soils were at least double the levels which would constitute acceptable risk. Cyanide levels were a concern for protection of wildlife but would not pose a risk to humans. Reducing exposure to cadmium to one half the present mean concentration, or to 250 mg/kg, would, based on the assumptions used in the risk evaluation and available data, reduce risk to

humans to a negligible level of 10^{-5}. Much lower exposure levels may be needed in order to protect wildlife, since cadmium is highly toxic to carnivorous wildlife through ingestion of contaminated dusts and invertebrate species, as well as potentially to fish if contaminants are dispersed off-site to Crab Orchard Lake. Cyanide residues are associated with sediments which contain cadmium, therefore, both contaminants will be addressed by the measures adopted to remediate cadmium. Remedial measures to address the complete transport pathways identified in the risk assessment will be given priority in the evaluation of alternatives for remedial response. Surface sediment dredging, capping, regrading, surface water diversion, and revegetating are among the technologies which will be applicable as part of this response.

The remedial objectives developed as part of the FS will be concerned with reducing risks to both humans and wildlife to acceptable levels and which will provide adequate margins of safety for protection of endangered wildlife.

Potentially applicable remedial measures for this site are discussed below.

Limited Site Access

It may be appropriate to take precautions to limit the potential for exposure to humans and wildlife via the water or direct contact with soils/sediments. Limiting site access could help prevent trespassing or unauthorized use of the site until the contaminated materials have been removed or contained. Fencing and closing the area to all but authorized personnel, and maintaining a thick vegetative cover may be appropriate until further remedial action can be initiated.

Surface Water Control

Drainage ditches or trenches might be required to divert surface water flow from areas containing elevated levels of site contaminants, in order to limit the off-site transport of contaminants.

Removal or Containment of Sediment

Contaminated surface sediments might be excavated and removed for treatment off-site or contained on-site. The contamination along the ditch extends approximately 5000 ft. towards the Lake, possibly 4 ft. wide x 2 ft. deep along the drainage route. The equivalent volume of contaminated sediment for excavation is estimated at 1600 CY. Clean fill will be used for regrading/fill for areas which may require excavation during remediation. Clean fill will be used for regrading/refilling excavated areas. Alternatively, the ditch could be drained and the sediments capped or sealed to prevent leaching of residual contaminants.

Monitoring - Surface and Groundwater

As part of the follow up remedial program, the response alternatives may include periodic sampling and analyses of the monitoring well and of the ditch water for cadmium, chromium and cyanide. Follow-up studies to verify the absence of contaminants might begin shortly after the cleanup.

29.6 Conclusions and Recommendations

It can be concluded that the Old Refuge Shop site is impacted with the primary pollutants being cadmium, cyanide, and chromium in the site's ditch water and sediment. It is recommended that remedial measures for this site should be evaluated in the FS. Potentially applicable remedial measures include limited site access to humans and wildlife and possible excavation of surface sediments.

SECTION 30 - SITE 24, PEPSI-WEST DRAINAGE

30.1 Site Description

Site 24 is a drainage ditch located north and west of the Pepsi-Cola Bottling Company building. The ditch runs parallel to the adjacent street. (See Figure 30-1). The ditch receives surface run-off from the site and discharges to Crab Orchard Lake. This site is not located on the Refuge and is not under the jurisdiction of U.S. FWS.

30.2 Site Investigations

30.2.1 Phase I Site Investigations:

One grab sample of surface water and one grab sample of sediment (0-1 ft depth) were collected from the drainage ditch.

30.2.2 Phase II Site Investigations:

One sediment sample was collected for mercury reanalysis.

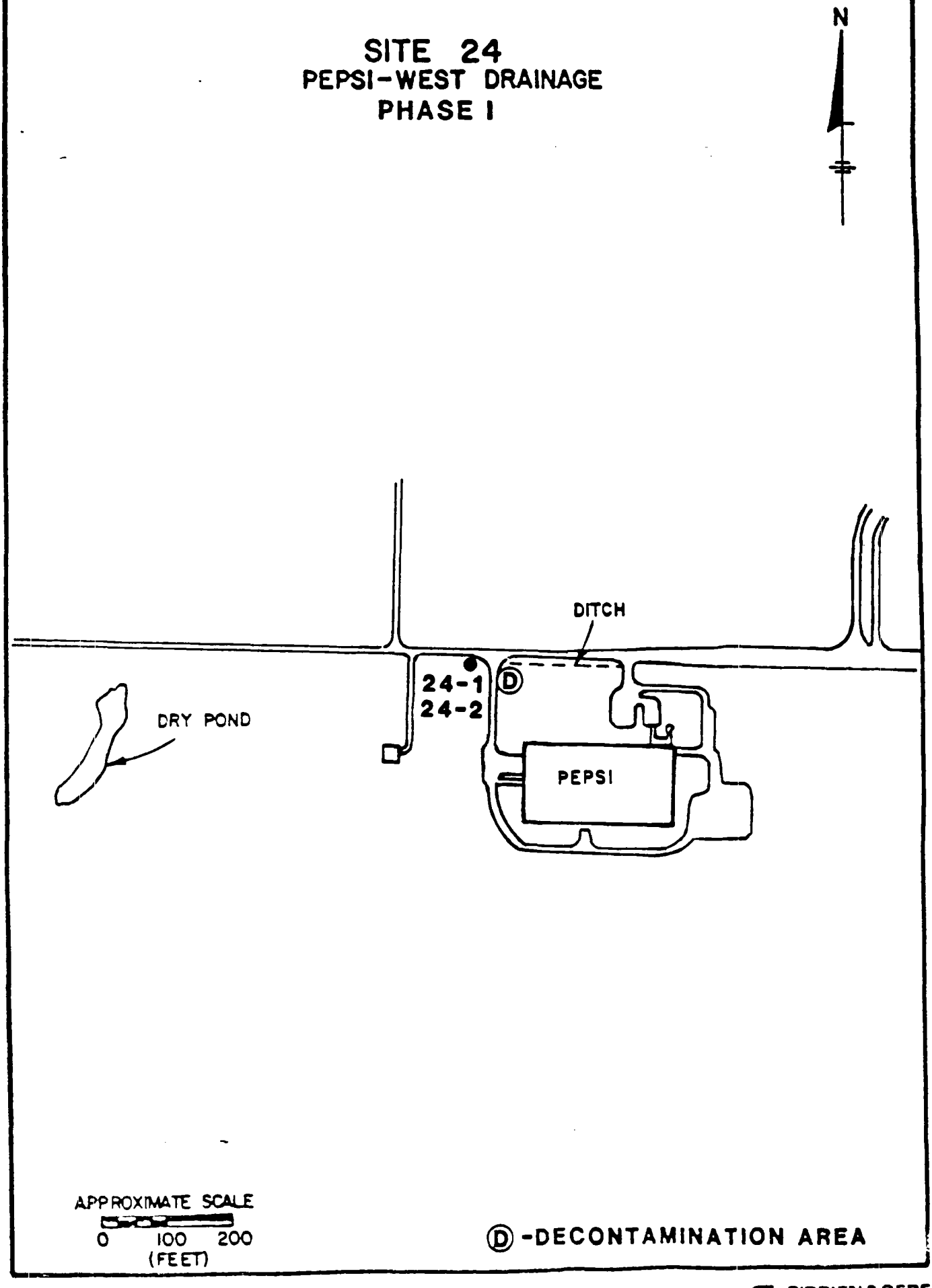
30.3 Analytical Results (See Appendix I, Page 23)

30.3.1 Phase I Analytical Results:

Total organic halide concentrations of 160 ug/L (190 ug/L duplicate) were detected in the water sample. The sediment contained acetone (268 ug/kg wet weight), and methylene chloride (117 ug/kg wet weight); however, these compounds were also detected in the laboratory blank. No other organics were detected in the sediment, although the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which

FIGURE 30-1

SITE 24
PEPSI-WEST DRAINAGE
PHASE I



O'BRIEN&GERE

were not detected may in fact be present. A mercury concentration of 9.4 ug/kg was detected, but this parameter was reanalyzed in Phase II due to questionable QA/QC support data.

30.3.2 Phase II Analytical Results:

The sediment sample contained 58 ug/kg of mercury. This concentration is greater than the background range measured at the control sites.

30.4 Environmental Effects

30.4.1 Qualitative Assessment

Sediments within the drainage ditch from Pepsi-West have been found to contain mercury at a level of 58 ug/kg. No other constituents detected in either water or sediments from this site were significantly above the levels found at the control sites. Available data do not indicate that surface water runoff from the ditch has transported mercury residues downstream to Crab Orchard Lake. Although the mercury concentration is somewhat higher than the level found at the control site soils, it is not unusual for mercury levels in U.S. soils (Lindsay, 1979). The detection of mercury at this site is not considered to be indicative of an off-site source of contamination. Since there is no waste source, a complete exposure scenario is not possible, thus it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

30.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified, there is not basis for preparing a quantitative risk evaluation.

30.4.3 Analysis of Uncertainties

The information relied upon for evaluating this location consisted of a site inspection and sample analyses. The chemical characterization data consisted of one surface water and two surface sediment samples from the ditch; deeper sediment cores or downstream water samples were not collected. Some characterization data are not sufficiently supported by QA/QC, and thus additional uncertainty is introduced dependent on the quality of the data. However, since there is no evidence to suggest that the site has been used presently or previously for disposal of wastes, and any potentially contaminated runoff from upstream areas would be transported via the surface water, the database should adequately characterize the site.

30.5 Preliminary Remedial Alternatives

The analytical results and the qualitative risk evaluation discussed in the previous sections indicate that this site does not contain contaminant levels that would result in a negative environmental impact. There will be no further evaluation of remedial alternatives, and this site will not be considered in the FS.

30.6 Conclusions and Recommendations

It can be concluded that the Pepsi-West drainage ditch site does not represent a risk of exposure to humans or to wildlife receptors. No further evaluation is recommended.

SECTION 31 - SITE 25, CRAB ORCHARD CREEK AT MARION LANDFILL

31.1 Site Description

The old Marion Landfill is located adjacent to Crab Orchard Creek on Old Creal Springs Road. This municipal landfill has been inactive for a number of years. A 3/4-acre pond is located next to the landfill. Site 25 consists of the Crab Orchard Creek sections upstream and downstream of the landfill and of the pond. (See Figure 31-1). This site is not located on the Refuge and is not under the jurisdiction of the U.S. FWS.

31.2 Site Investigations

31.2.1 Phase I Site Investigations:

One composite surface water sample and one composite sediment sample (0-1 ft depth) were collected at each of two locations approximately 250 ft. upstream and downstream of the landfill. An additional downstream sediment was also collected for CLP organics analysis. A composite surface water sample and a composite sediment sample (0-1 ft depth) were also collected from the pond.

31.2.2 Phase II Site Investigations:

One of the creek sediment samples was resampled for cyanide analysis.

31.3 Analytical Results (See Appendix I, Page 24)

31.3.1 Phase I Analytical Results:

Magnesium in the water increased from 14.3 mg/L in the upstream water sample to 47 mg/L in the downstream sample. Manganese in the

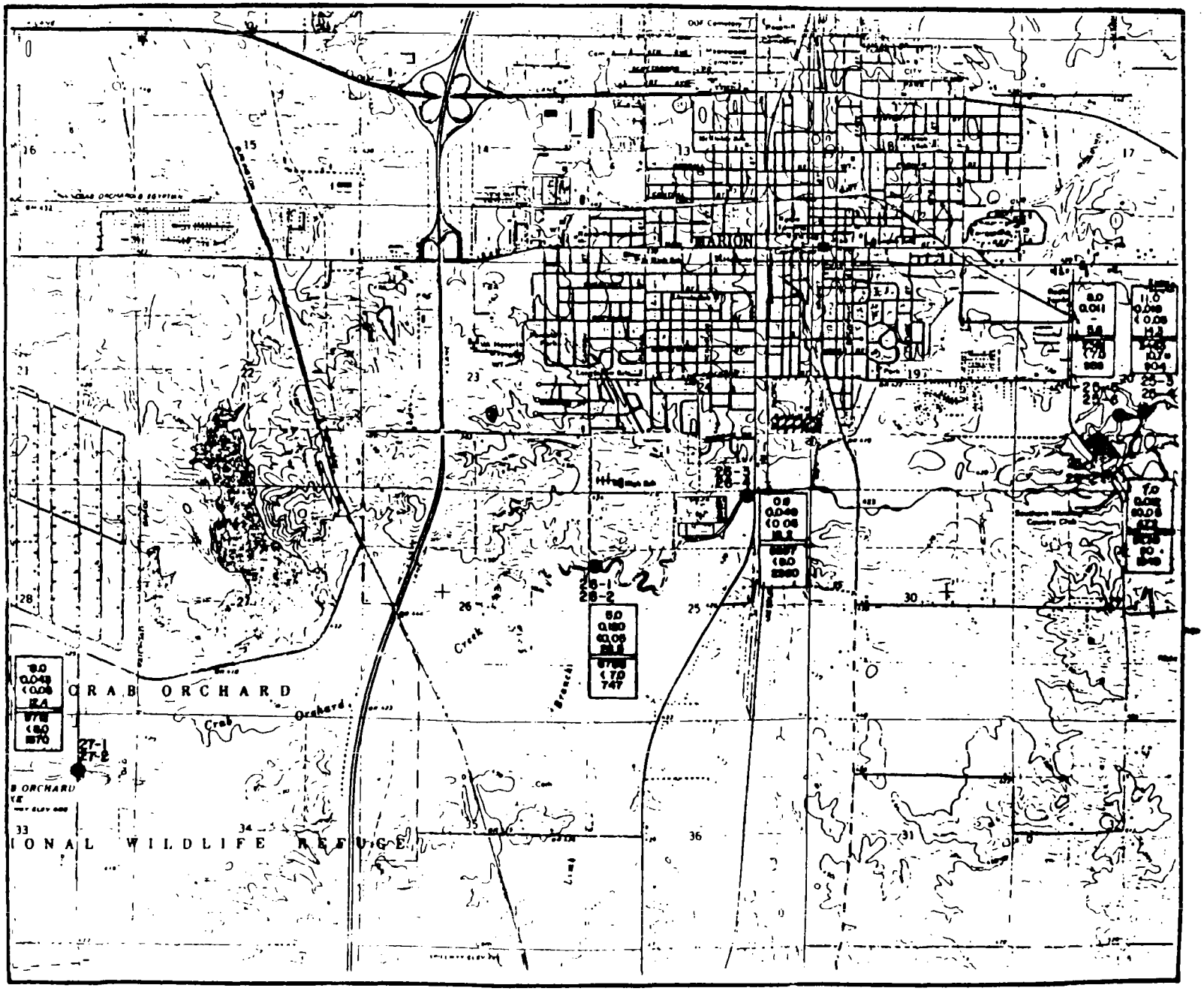


FIGURE 31-1

N

SITES 25, 26, & 27
CRAB ORCHARD CREEK

PHASE I

LEGEND

CONCENTRATIONS

WATER, mg/L

SEDIMENT, mg/kg

- PHASE II REANALYSIS

- MARION SEWAGE TREATMENT PLANT

- APPROXIMATE LOCATION OF OLD MARION LANDFILL

SCALE IN FEET

0 2000 4000

upstream water (0.68 mg/L), in the downstream water (1.5 mg/L) and in the pond (0.72 mg/L) exceeded the Federal MCL and State standard of 0.50 mg/L, but all other contaminant levels in waters were below the Illinois Public Water Supply Standards. The upstream water and pond water also exceeded the Federal MCL for iron but were at or below the Illinois Public Water Supply standard. The iron and manganese concentrations in water are not considered to pose health risks to human populations or wildlife under these conditions (see Exhibit A), since their standards are based on concerns of taste and color.

In general, the concentrations of most constituents increased in the downstream samples. Figure 31-1 shows concentrations for total organic carbon (TOC), total organic halides (TOX), cyanide and magnesium along the creek. It is noted that cyanide values are questionable due to poor QA/QC data and metals concentrations are reported as estimated values only. The creek sediments at Site 25 contained similar levels of magnesium (904 mg/kg upstream and 1,840 mg/kg downstream) as detected at the control sites. The downstream sediment contained cyanide (90.4 mg/kg) compared to below 8 mg/kg upstream, and elevated TOC (18,239 mg/kg) compared to 3,778 mg/kg upstream. The pond sediment contained magnesium (956 mg/kg), and below 7 mg/kg cyanide. No priority pollutants were detected in the CLP analysis, although FID scans in the sediments ranged from 3,735 ug/kg to 220,368 ug/kg.

31.3.2 Phase II Analytical Results:

The cyanide concentration in the creek sediment upstream of Site 25 was 10.7 mg/kg.

31.4 Environmental Effects

Environmental Effects for sites along Crab Orchard Creek are discussed in Section 33.4.

31.5 Preliminary Remedial Alternatives

Preliminary Remedial Alternatives for sites along Crab Orchard Creek are discussed in Section 33.5.

31.6 Conclusions and Recommendations

Conclusions and Recommendations for sites along Crab Orchard Creek are discussed in Section 33.6.

SECTION 32 - SITE 26, CRAB ORCHARD CREEK BELOW MARION STP

32.1 Site Description

The Marion Sewage Treatment Plant (STP) discharges to Crab Orchard Creek upstream of Court Street in the Village of Marion. This site is not within the boundaries of the Refuge and is not under the jurisdiction of the U.S. FWS.

32.2 Site Investigations

32.2.1 Phase I Site Investigations:

One composite surface water sample and one composite sediment sample (0-1 ft depth) were collected at each of two locations, both downstream of the treatment plant, to assess the impact on various segments of the creek. The first sampling location was at the intersection of the creek and So. Carbon Street (samples 26-1, 26-2). The second sampling location was at the intersection of the Creek and Court Street (samples 26-3, 26-4). The sampling locations were spaced approximately 2,000 ft. apart (see Figure 31-1).

32.2.2 Phase II Site Investigations:

No samples were collected in Phase II.

32.3 Analytical Results (See Appendix I, Page 25)

The FID scan, lead, magnesium, and zinc concentrations were approximately one order of magnitude higher in the downstream sediment samples than in the upstream samples, but all concentrations were similar to those detected at the control sites. The metals concentrations are included as

estimated values for screening purposes only (see Exhibit B). Chloroform was detected in both the upstream and downstream water samples at 2 ug/L, exceeding the AWQC for human health of 0.19 ug/L, but is well below the same criterion set for aquatic life protection of 1,240 ug/L. No other volatile organics or PCBs/Pesticides were detected in either the water or sediments although TOX and FID scans detected organics. Semivolatiles were not analysed at this site.

TOX levels were somewhat elevated in the downstream water sample, 120 ug/L (130 ug/L duplicate) versus 49 ug/L upstream. Manganese concentrations in the upstream and downstream waters (0.3 and 0.75 mg/L) both exceeded the Federal MCL and Illinois Public Water Supply standards. Iron in the downstream sample (1.0 mg/L) also exceeded the Federal MCL, but not the Illinois standard. Iron levels are regulated due to objectionable taste and color; at these levels, this does not pose a threat to public health. All other surface water parameters were within Illinois Public Water Supply Standards. Figure 31-1 shows the concentration for TOC, TOX, cyanide and magnesium in waters and sediments from Crab Orchard Creek. All parameters in the upstream location (26-1, 26-2) were below the levels found upstream of the STP (Site 25), with the exception of TOX.

32.4 Environmental Effects

Environmental Effects for sites along Crab Orchard Creek are discussed in Section 33.4.

32.5 Preliminary Remedial Alternatives

Preliminary Remedial Alternatives for sites along Crab Orchard Creek are discussed in Section 33.5.

32.6 Conclusions and Recommendations

Conclusions and recommendations for sites along Crab Orchard Creek are discussed in Section 33.6.

SECTION 33 - SITE 27, CRAB ORCHARD CREEK BELOW I-57 DREDGE AREA

33.1 Site Description

Site 27 is located in Crab Orchard Creek downstream of the Interstate Route 57, approximately 4,000 ft. downstream from Site 26 (See Figure 31-1). Dredging of the stream bed was conducted in this area a number of years ago. This site is located on the Refuge.

33.2 Site Investigations

33.2.1 Phase I Site Investigations:

A composite surface water sample and a composite sediment sample (0-1 ft depth) were collected from Crab Orchard Creek where the creek intersects with Chammness Road.

33.2.2 Phase II Site Investigations:

No samples were collected in Phase II.

33.3 Analytical Results (See Appendix I, page 26)

TOX levels in the surface water of 43 ug/L (38 ug/L duplicate) were below those found at Site 26. Most other parameters were similarly below the levels detected at Site 26 except for TOC and magnesium in the sediment. Figure 31-1 shows the concentration for TOC, TOX, cyanide and magnesium along Crab Orchard Creek. The cyanide results were questioned due to poor QA/QC and the metals concentrations are estimated values (see Exhibit B). The manganese concentration (0.64 mg/L) in the water exceeded the Federal MCL and Illinois Public Water Supply Standard as it did in each of the five samples collected further upstream. Iron (0.5 mg/L) also exceeded the Federal

MCL but not the Illinois standard. The concentrations of all other contaminants in the water were below the Illinois Public Water Supply Standards and Federal MCLs. The excursions noted for iron and manganese do not represent a health concern since these standards are implemented based on aesthetic considerations of taste and color.

Sediment concentrations were similar to the ranges detected for the Refuge control sites.

33.4 Environmental Effects

33.4.1 Qualitative Assessment

33.4.1.1 Source Evaluation

This section is an analysis of exposure and risk in Crab Orchard Creek at the Marion Landfill (Section 31), below the Marion Sewage Treatment Plant (Section 32), and below the I-57 Dredge Area (Section 33).

33.4.1 Qualitative Assessment

33.4.1.1 Source Evaluation

Crab Orchard Creek At Marion Landfill

Crab Orchard Creek may receive leachate and runoff from the Old Marion Landfill, now inactive. Analyses of the site revealed magnesium residues up to 47 mg/L in the creek water and up to 1,840 mg/kg in the sediment of a nearby pond; iron concentrations in water up to 1,000 ug/L, manganese levels in water up to 1,510 ug/L, and cyanide in creekbed sediment of 10.7 to possibly 90.4 mg/kg were also found. Exceedance of the standards for manganese and iron do not represent a health concern since these criteria were established based on aesthetic considerations.

The reported maximum cyanide concentration of 90 mg/kg is estimated due to deficiencies in QA/QC support data from the Phase I analysis. Since no other water or sediment samples along Crab Orchard Creek contained elevated levels of cyanide, this value seems to represent a false positive, and cyanide will not be considered as a site contaminant. Magnesium levels are within the ranges typically found in soils (Lindsay, 1979). Due to the lack of source of exposure, a complete scenario is not possible and this site will not be considered further.

Crab Orchard Creek Below Marion Sewage Treatment Plant

No water or sediment cyanide levels were detected at this site that would constitute a threat to human health or the environment. Iron and manganese levels in water were above the aesthetic-based standards but are not considered to represent a health concern. Due to the lack of source of exposure, this site will not be considered further.

Crab Orchard Creek Below I-57 Dredge Area

As discussed in Section 33.3, water sampled from the creek contained iron and manganese levels above the Illinois Public Water Supply and Federal drinking water standards. Iron and manganese levels were above the MCL set for aesthetic concerns but were within the standards for health protection. Sediment levels were comparable to the Refuge control sites. Magnesium in the creek water ranged from 5.6 to 47.2 mg/L. Since there is an insufficient data base from which to determine risks to fish, wildlife, and humans at this site, and since federal criteria or standards for magnesium have not been promulgated, a receptor analysis and quantitative risk assessment will not be performed.

33.4.3 Analysis of Uncertainties

Several areas of uncertainty can be noted in the evaluation of Sites 25, 26, and 27 along Crab Orchard Creek:

1. The limited number of samples collected may not completely characterize the sites, given the large geographical area studied;
2. The uncertainties inherent in some of the laboratory analyses due to inadequate QA/QC support data may underestimate the presence of site contaminants;
3. The chemical form and toxicity of cyanide residues, if any, have not been determined.

33.5 Preliminary Remedial Alternatives

Follow-up monitoring of surface water and sediments is suggested. Attachment 1 details a recommended monitoring program. The parameters identified as possible targets for monitoring along Crab Orchard Creek sites include total organic carbon, total organic halides, cyanide, manganese, iron, and magnesium.

33.6 Conclusions and Recommendations

The water samples from Crab Orchard Creek exceeded the State and Federal standards for iron and manganese, however these standards were established based on aesthetic concerns of taste and color, therefore, the levels found do not pose a threat to human health or wildlife. Sediment contaminant levels detected were not supported by QA/QC data and do not appear to have leached to the water. It can be concluded that follow-up monitoring studies (see Attachment 1) will adequately characterize the conditions of the site in the event that these conditions change.

SECTION 34 - SITE 28, WATER TOWER LANDFILL

34.1 Site Description

Historical aerial photographs indicate that landfilling activities occurred at Site 28, adjacent to the Water Tower near Areas 7 and 14. (See Figure 34-1).

The photographs from 1943 show that the area encompassed by the Water Tower road is radially marked by tire tracks, and debris can be seen at the end of the tracks. The landfill appears as a diamond-shaped area to the north of the Water Tower. Photographs from 1951 show that the landfill was not in use at that time, and the area previously noted to be marked with tire tracks is now vegetated and the landfill area is no longer distinguishable. The landfill appears to have been reactivated sometime after 1951, as debris and equipment are shown in aerial photographs from 1960 and 1965. By 1971, the landfill is no longer active and the area is partially vegetated. The areas of site activity indicated on the 1943 and 1965 air photos are illustrated on Figure 34-1.

These activities are not visually apparent today; however, a number of rusted drums, metal parts and tar residues are present. The site gradually slopes to the northeast. The sloping face northeast of the Water Tower is heavily overgrown with briars and rutted with several major gullies; only a small amount of refuse is evident in this area. More activity is evident in the woods at the bottom of the slope. Standing water in the main drainage gully showed a slight sheen on the surface on one site inspection, but was not evident in latter visits to the site. This gully ultimately discharges approximately 1 mile northward to Crab Orchard Lake. Several small mounds are within the woods and a larger mound is located at the top of the hill. Previous soil sampling by DOI detected lead concentrations up to 800 mg/kg. (Ruelle, February 1983).

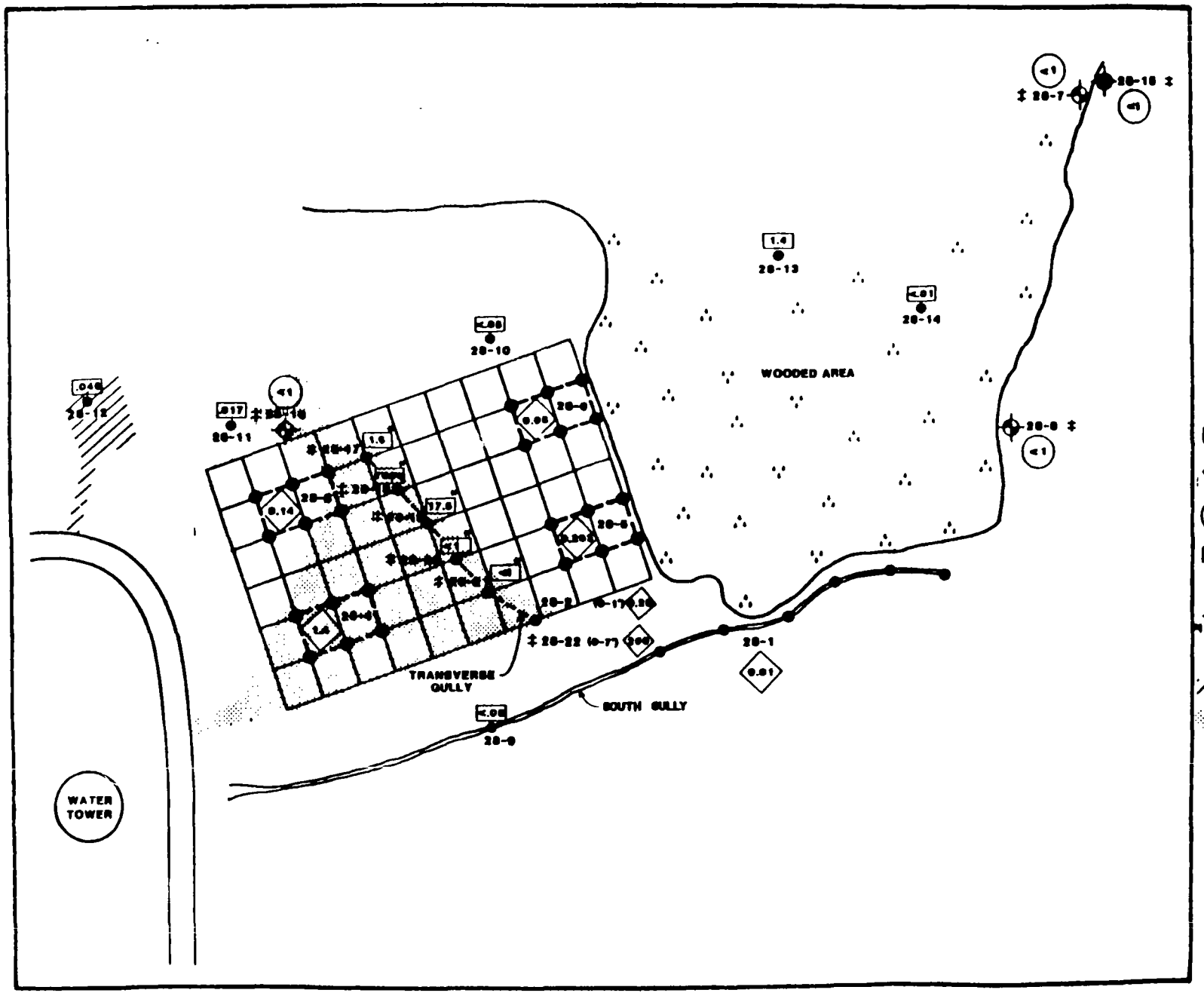


FIGURE 34-1

N

SITE 28
WATER TOWER LANDFILL
PHASE I & II

LEGEND

PCB CONCENTRATIONS

GRAB SAMPLE SOIL, mg/kg WET WEIGHT

AREAL SURFACE COMPOSITE SOIL, mg/kg WET WEIGHT

WELL, mg/L

SOIL, mg/kg WET WEIGHT 0-7' TEST PIT

SHALLOW WELL

DEEP WELL

‡ PHASE II SAMPLES

/// - AREA OF SITE ACTIVITY INDICATED ON 1943 AIR PHOTO

▒ - AREA OF SITE ACTIVITY INDICATED ON 1965 AIR PHOTO

SCALE IN FEET

0 50 100

34.2 Site Investigations

34.2.1 Phase I Site Investigations:

A magnetometer and electromagnetic terrain conductivity survey was conducted along and transverse to the slope of the landfill. (See Figures 34-2 and 34-3). Two shallow ground water monitoring wells were also installed and set at depths of 20 and 25 feet in clayey silt. Both wells included ten foot length well screens from intervals of 10-20 and 15-25 feet respectively.

A survey grid was established for the area northeast of the Water Tower where burial activities may have occurred based on the aerial photographs from 1965. Composite soil samples each consisting of six, 0-1 ft. depth grabs) were collected along a central transverse drainage gully, the Water Tower drainage ditch (south of the grid), and from grids spaced toward the outer edges of the grid. Six additional grab soil samples (0-1 ft. depth) were collected from other locations, including two from the diamond shaped area to the north of the Tower, two within the wooded area where scattered debris was found, and two from locations removed from the grid. One sample from the transverse gully was resampled for full CLP organics analyses.

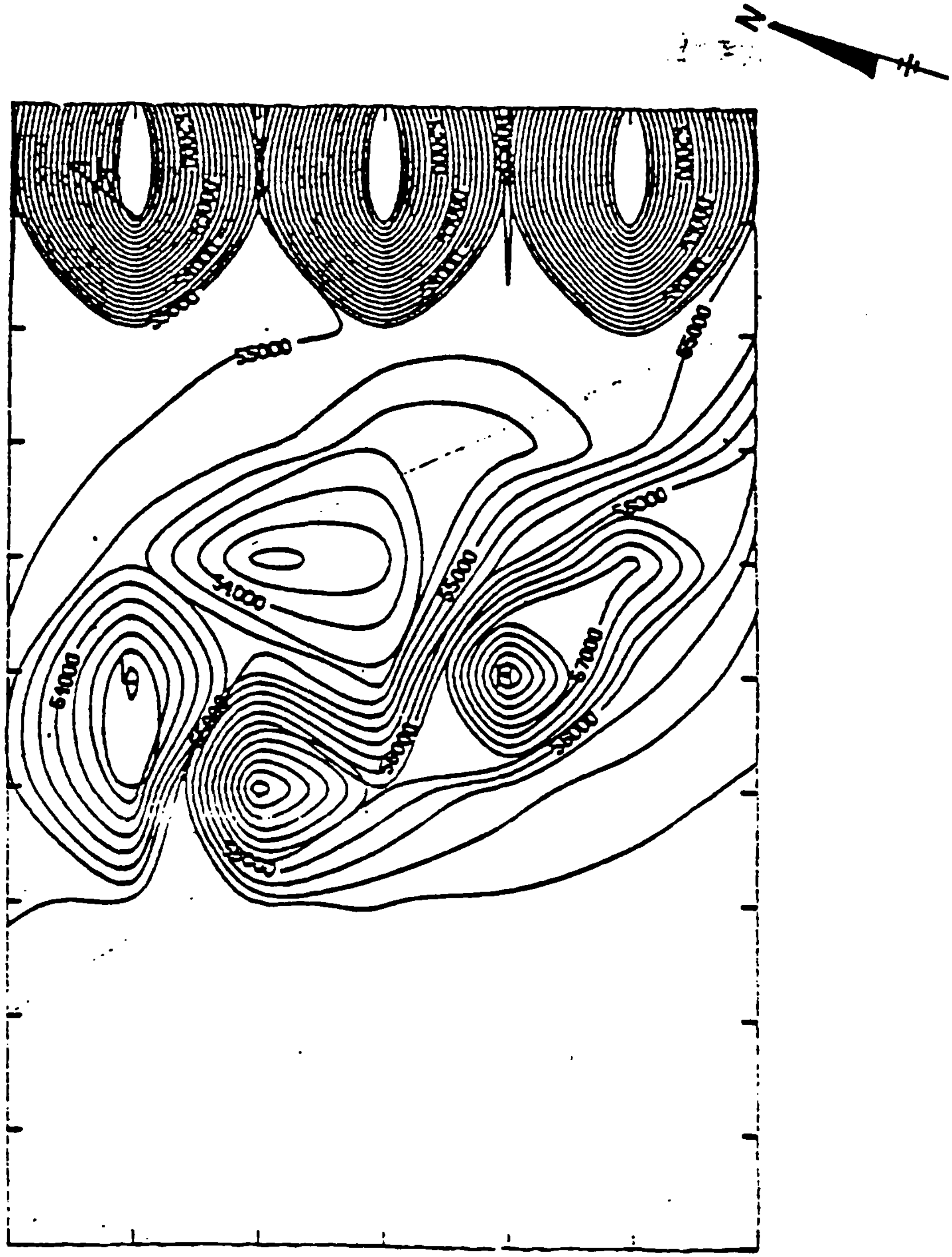
34.2.2 Phase II Site Investigations:

Phase II site investigations included the installation and sampling of a set of nested ground water monitoring wells in addition to the sampling of the two wells installed in Phase I. The Phase II wells were set at depths of 20.5 and 39 feet and screened from 15.5-20.5 and 34-39 feet respectively. Soils encountered at these sites were generally silt and silty clay with a fine sand layer identified at 34-39 feet in the deep well. Well

FIGURE 34-2

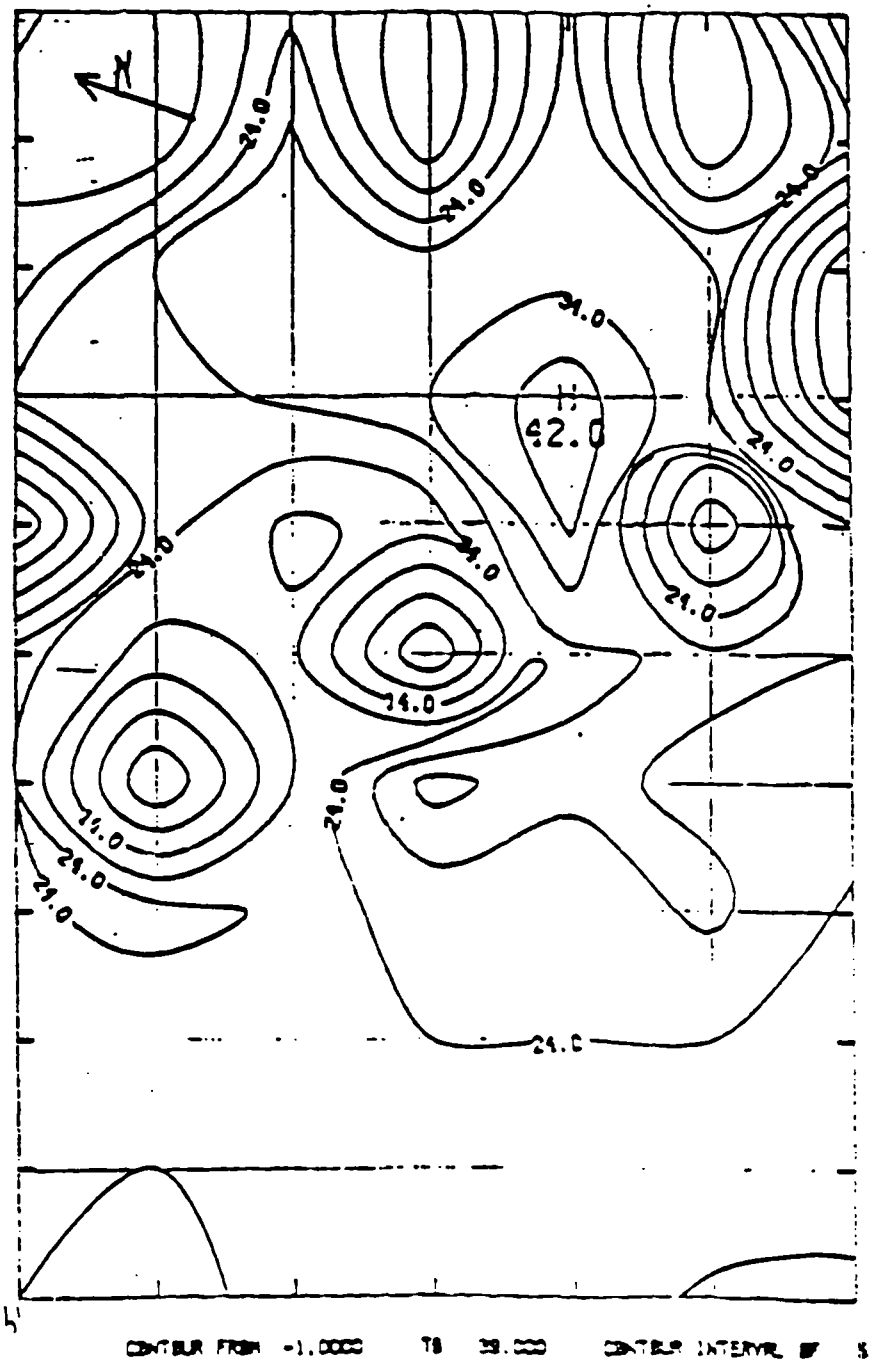
SITE 28

MAGNETOMETER SURVEY



SITE 28

ELECTROMAGNETIC SURVEY



28-15. The ground water samples were analyzed for CLP HSL volatiles, pesticides, PCBs, and metals, as well as for cyanide and indicators.

Five soil test pits were dug to a depth of 7 feet. The test pit locations were selected based on the data results from the geophysical surveys. The pits were dug to explore subsurface soils, although the Phase I screening did not show elevated contaminants in any of the surface soils. Six soil samples were collected from these pits: one composite (0-7 ft. depth) from each pit, as well as one composite combining the five test pit samples (See Figure 34-1). The soils were analyzed for PCBs, magnesium, copper, lead, arsenic, and cyanide.

34.2.3 Site Hydrogeologic Characterization

34.2.3.1 Site Geology

Based on results of the well installations at Borings 28-7, 8, 15 and 16, the subsurface unconsolidated overburden consists of a mottled grey, orange, and brown silty clay to clayey silt containing trace amounts of sand and fine gravel. This material continues vertically to depths of 20 to 25 ft. below ground level, and appears to be laterally continuous throughout the site. Beneath the silty clay layer, approximately 12 ft. of clay is present to a depth of 24 to 36 ft. as exhibited in the deep boring 28-15. A thin layer of fine-coarse sand is present beneath the clay, and occurs above a medium grained, light grey sandstone bedrock. Top of bedrock is 37.5 ft. below ground level in the deep boring. As only one boring encountered materials beneath the silty clay layer, no estimate can be made of the lateral extent of the lower unconsolidated layers.

34.2.3.2 Site Hydrogeology

Occurrence of Ground Water

Shallow ground water occurring beneath the site was found at a depth of 1.5 to 18 ft. below the ground surface within the silty clay/clayey silt soil unit. The three shallow ground water monitoring wells were screened in this upper water table. Ground water elevations collected during the winter and summer of 1987 (wet and dry seasons, respectively) indicate a water table fluctuation of 2 to 4 ft. with water levels dropping during the summer months (Table 4-3).

A lower ground water aquifer was encountered in the lower portions of the soil sequence on top of the sandstone bedrock where unconsolidated sands were encountered. The deep well installed on site was screened in this lower aquifer. Ground water elevations collected during the winter and summer of 1987 indicate a 0.75 ft. fluctuation in the water table with levels dropping in the summer months (Table 4-3). This lower aquifer is confined, as is apparent by a 3 ft. higher elevation in ground water in the deep well as opposed to that occurring in an adjacent nested shallow well.

Ground Water Flow Conditions

Ground water elevations from the shallow ground water monitoring wells were contoured and are presented in Figure 34-4. The ground water flow direction is toward the northeast, i.e. along a north-south trending stream which flows north into Crab Orchard Lake. The hydraulic gradient of flow (i) during June 18-19, 1987 was approximately 0.016 ft/ft. The average hydraulic conductivity (K) for the shallow wells was calculated to be about 0.37 ft/day. Porosity was assumed to be 0.35 (Davis and Dewiest).

A calculation was then made of the average ground water flow velocity (Vs) through the upper soil units. Using the formula given in Section 4.2, the resultant velocity was calculated to be about 0.017 ft/day or 6.2 ft/year. The flow velocity is controlled by the relatively low hydraulic gradient and low hydraulic conductivity occurring in this area. An upward vertical flow was identified from the unconsolidated aquifer screened by the deep Well 28-15 into the upper water table. This phenomenon indicates discharge of deep ground water towards Crab Orchard Lake.

34.3 Analytical Results (See Appendix I, Page 27)

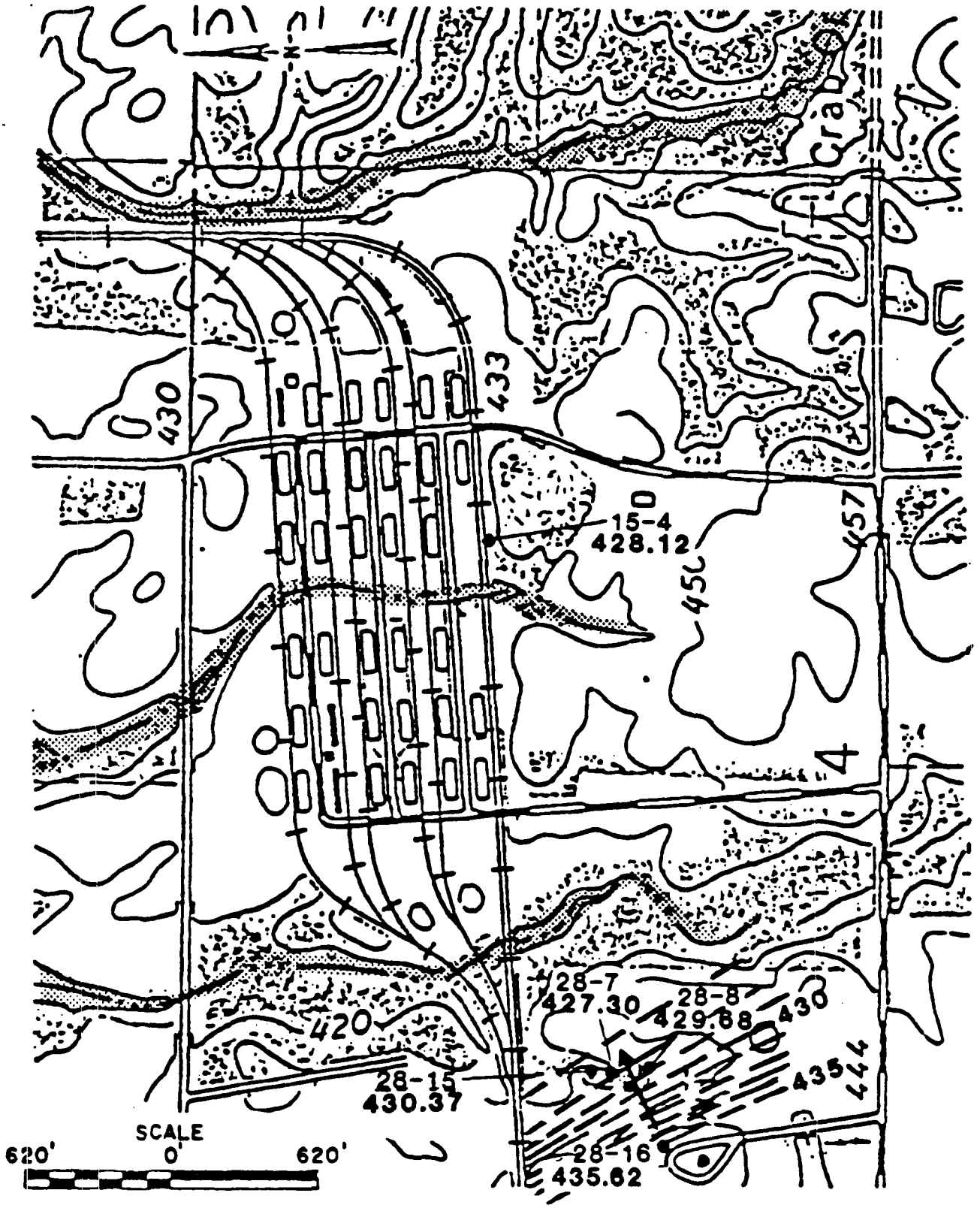
34.3.1 Phase I Analytical Results:

The magnetometer and electromagnetic surveys shown in Figures 34-2 and 34-3 suggested the presence of metallic debris within the transverse ditch. Some scattered residues of metallic debris were found during the site inspection within the wooded area to the east.

The symmetrical contour lines shown in Figure 34-2 north east of the grid resulted from the more narrow grid spacing which had to be used within the wooded area, and was not due to any metallic or conductive objects at the Site. Figure 34-1 shows the detected PCB concentrations on a wet weight basis. Two of the soil samples (28-4, a surface grid composite, and 28-13, a grab from the wooded area) contained 2.8 and 1.7 mg/kg dry weight respectively, which were higher than those detected in the other soils (less than 0.01 to 0.354 mg/kg dry weight). The soil composite collected from the transverse gully (sample 28-2) at a 0-1 ft depth contained 0.35 mg PCBs/kg. The soil composite along the ditch south of the landfill area contained 0.023 mg PCBs/kg dry weight.

FIGURE 34-4

SITE 15 & 28 GROUNDWATER FLOW MAP



SITE 15 & 28
15 - ACID POND
18 - WATER TOWER
▒ - STREAM

28-7 JUNE 18, 1987
● - MONITORING WELL LOCATION & IDENTIFICATION
427.30 - GROUND WATER ELEVATION
430 — — — GROUND WATER CONTOUR

O'BRIEN & GERE ENGINEERS INC

Lead was detected in all of the surface soil samples at concentrations ranging from less than 20 to 250 mg/kg. Lead concentrations are estimated and are reported for screening purposes only. The highest lead concentration was detected at Location 28-4, which also contained the highest PCB concentration. The range of lead concentrations measured would not be considered atypical for lead levels in soils.

34.3.2 Phase II Analytical Results:

The soil samples collected along the transverse gully (Figure 34-1; location 28-2, 0-1 ft composite for Phase I, and locations 28-17 to 28-22 at 0-7 ft depth for Phase II), contained PCB concentrations in the range of less than 1 to 22 mg/kg for four test pits, and one high concentration of 8,900 mg/kg at test pit location 28-18. The composite of the five test pits (Sample 28-22) contained 320 mg/kg PCBs, skewed by the detection of residues in one test pit. Lead concentrations ranged from 13 to 712 mg/kg for the test pit soils with the exception of sample 28-18 which contained 4,300 mg lead/kg. Copper concentration was also elevated (8.4 to 813 mg/kg) compared to the Refuge background. The highest copper value was detected in the sample from 28-19, adjacent to the pit with elevated PCBs and lead. Magnesium levels were found at 2,940-16,300 mg/kg which are within an order of magnitude of the Refuge background.

The ground water samples all contained below 1 ug/L PCBs. Traces of chloroform were detected in nested wells 28-15 and 28-16 only, at 2 and 10 ug/L, respectively. Acetone and methylene chloride were reported in the waters but their presence is attributed to laboratory contamination. No pesticides were detected.

All dissolved metals concentrations were below the Illinois Public Water Supply Standards and Federal MCLs and MCLGs; however, only the unfiltered samples were analyzed for wells 28-15 and 28-16. The concentrations of some metals were detected above the standards for these unfiltered well samples. Total copper was 117 ug/L in Well 28-16, compared to the Federal Drinking Water standard of 1,000 ug/L, and the Illinois State standard of 20 ug/L. Manganese exceeded the Illinois and Federal MCL standard of 50 ug/L for the same well at a concentration of 2,780 ug/L total. Total iron in the monitoring wells exceeded the Federal MCL standard for iron of 300 mg/L, with concentrations between 425 ug/L and 94,600 ug/L. It should be noted that the excursions for iron and manganese are not considered to represent a concern for public health or wildlife protection, since these standards are promulgated due to considerations of taste and color. Unfiltered lead (4.5-76 ug/L) and zinc (16-355 ug/L) were also elevated, but only lead was above the Federal standard of 50 ug/L.

34.4 Environmental Effects

34.4.1 Qualitative Assessment

34.4.1.1 Source Evaluation

Although aerial photographs indicate that a waste disposal area existed in the vicinity of the Water Tower, there is no information currently available regarding the history or source of the wastes disposed of at this site. Most information regarding the nature and composition of wastes in this area is based on the test pit investigation. The test pit investigation, which was based on the results from the geophysical surveys, revealed that the metallic objects in the landfill were not

directly identifiable as being of municipal origin, but appeared to be small pieces of stamped metal, wire and other objects. It is presumed that these materials may have originated from one or more of the industrial sites in the area.

There were no containers or containerized wastes located in any of the test pits, nor were any such materials located on the surface of the site, with the exception of one empty rusted drum in the wooded area. Likewise, there were no liquid or semi-liquid wastes, such as oils or tars, observed in the test pits or surfaces of the site. The primary waste materials encountered were what appeared to be wires, small metal parts and small electrical components, such as switches. There were no solid chemical wastes observed in the test pits or at the surface of the site. At this time, with the exception of the test pit areas, the site is covered by vegetation.

Based on the results of the surface soil sampling, the surface of the site appears to be free from waste contamination. The results of the test pit investigation indicate the presence of PCBs and lead at one of the subsurface locations. With the exception of PCBs, priority pollutant organics were not significantly different from those detected at the control sites.

Although all of the components of waste at this site are not known, based on the chemical compounds detected (PCBs, lead and copper), the waste residues are not especially mobile, and would therefore be expected to remain in place within the landfilled area. Due to low vapor pressures characteristic of these compounds, these compounds do not tend to volatilize. Also, PCBs have a very low solubility in water and a high organic carbon adsorption coefficient. These two factors significantly retard the leaching of PCBs into the ground water.

Lead and copper also have a limited potential for mobility. Depending on the type of compound they are a component of, these two metals demonstrate only low to moderate solubilities in water. Also, as cations, they become adsorbed to active (predominantly negative) surfaces on soils and sediments, thereby retarding their ability to be leached and be transported with ground water.

Of the materials detected within the site, PCBs have the highest potential to cause toxic effects. PCBs have been demonstrated to cause both acute and chronic toxic effects at low doses. Acute effects are of most concern to fish and wildlife (Exhibit A). PCBs have also been demonstrated to be carcinogenic in animal studies. In comparison with the toxic properties of PCBs, lead and copper are considered as minor toxic components at the levels at which these metals were detected at the site. The PCBs will therefore be the major source compound considered in this risk assessment.

34.4.1.2 Transport Route Evaluation

a) Air: Because the waste materials are currently covered by a layer of vegetation and have not been shown to be present in samples of surface soils, the waste materials present at this location are not considered to be a significant source for release to and transport by the air route. The site contaminants (PCBs, lead and copper) do not exert an appreciable vapor pressure and will not diffuse out of soils into air at significant concentrations. For these reasons, the air route is concluded to be non-functional at this location.

b) Direct Contact: As with the air route, the direct contact route cannot be considered to be complete, based on the absence of

exposed waste materials. However, in the event that portions of the site cover are disrupted, such as by excavation or by burrowing activities by small mammals, wastes may become exposed, thus creating a source for the direct contact pathway. However, in its present condition it can be assumed that the direct contact route is not functional at this location.

c) Surface Water: Based on the absence of exposed waste materials that could come in contact with surface water runoff at or near the site, it can be concluded that the surface water transport route is not functional at this location. However, should the waste materials become exposed, they could be scoured by precipitation and runoff, thereby entering the surface water.

d) Ground Water: The results of the groundwater sampling and analysis program indicate that waste components are not present in groundwater above the limits of detection. The measured soil hydraulic conductivity at this site ranges from 2.48×10^{-7} to 5.93×10^{-6} ft/sec (Table 4-4). The transport of PCBs and lead are significantly retarded by this type of soil. On this basis, and the results of the ground water analyses which indicate the absence of waste components, it can be concluded that the ground water transport route is not functional.

34.4.1.3 Receptor Evaluation

Human

Based on the results of the site inspection and analytical program, and the fact that the waste materials are currently covered by soils and vegetation, the transport route evaluation indicated that there is

currently no potential route by which human receptors in the area may become exposed to the waste materials. However, should this area be excavated in the future for construction or other purposes, all of the routes, with the exception of the groundwater route, would become functional and the potential for human exposures would arise.

Wildlife

The Water Tower Landfill is in a wildlife refuge and therefore the potential for contaminant exposure is also possible for wildlife. As with human exposures, the fact that the waste materials are currently covered with vegetated topsoil generally provides a barrier against exposures by wildlife. However, exposures may be experienced by burrowing animals if these were to establish dens within the area of the fill. Nevertheless these exposures would be minimal, based on the results of the test pit investigation, which identified PCBs and metals in isolated areas, rather than evenly distributed throughout the site.

34.4.2 Quantitative Assessment

Because no complete exposure scenarios could be identified in the qualitative risk assessment, there is no basis for preparing a quantitative risk evaluation.

34.4.3 Analysis of Uncertainties

The sampling locations for this site were based on reviews of historical aerial photographs, geophysical surveys, and site inspections. This information was used both for selection of surface and deep soil sampling locations. Based on the results of these surveys, in addition to

the Phase I and Phase II analytical sampling programs for soils and ground water at this site, it is concluded that the data generated should be adequate to characterize the site.

34.5 Preliminary Remedial Alternatives

The Phase I and Phase II sampling program for this site did not detect surface soil contamination or contaminant migration via ground waters. All constituents analyzed in the surface samples were within the ranges found at the Refuge control sites. Deeper soil samples (1 to 7 ft. depths test pits) from one area revealed a localized source of PCBs and lead. However, based on the overall evaluation of site conditions, hydrogeology, ground water analyses, surface and subsurface soil characteristics, the conclusions of the risk evaluation are that the waste source is adequately isolated, and no complete pathways were identified that might pose a risk to wildlife or humans under present site conditions.

Based on the conclusions derived from the risk evaluation, potentially applicable remedial efforts to be evaluated in the FS might include ground water monitoring, limited site access, capping, surface water diversion, regrading, and revegetating. Site characterization data generally support these remedial technologies; however, in the event that excavation and transport of materials is considered, additional sampling might be conducted to more accurately define the waste source. Estimates for evaluation of remedial response alternatives in the FS will be made based on available data including aerial photographs, geophysical surveys, and the RI sampling results.

A summary of potential remedial actions for this and all sites at the Refuge is shown in Table 2 of the Executive Summary. The remedial technologies will be reviewed in more detail as part of the FS.

Any considerations for future land uses should be evaluated and approved by the Refuge Management. Some of the potentially applicable remedial measures for this site are reviewed below.

Limited Site Access

It may be advisable to reduce human and/or wildlife access to the Water Tower site. Construction of a fence around the landfill area would accomplish this, although maintaining a thick vegetative cover may provide adequate protection since wastes are currently not exposed. Deed restrictions may be incorporated to control future uses of the site.

Capping, Regrading, Control of Surface Water

To further limit the potential for exposure to subsurface contamination, engineering controls such as capping may be instituted at this site. Regrading and diversion of surface streams and drainage channels may be implemented in lieu or in conjunction with the cap, to protect surface soil erosion.

Monitoring - Groundwater

The remedial response actions may include periodic sampling and analyses of the four monitoring wells for PCBs and lead.

34.6 Conclusions and Recommendations

It can be concluded that the Water Tower Landfill site contains subsurface soil contaminants with the primary pollutants being PCBs and lead. It is recommended that remedial alternatives for this site be evaluated in the FS. Since the sampling results confirmed only subsurface contamination, remedial efforts will focus on preventing future exposure to human or wildlife receptors. Potentially applicable remedial measures to be evaluated in the FS include regrading, surface water diversion, drainage controls, capping, limited site access, and ground water monitoring.

SECTION 35 - SITE 29, FIRE STATION LANDFILL

35.1 Site Description

Site 29 is a large open field (roughly 350 ft. x 300 ft) southwest of the Refuge Fire Station. (See Figure 35-1). The field was used for storage of mining machinery until several years ago. Prior to that, the landfill was reportedly used by Olin, and a fire is reported to have occurred. Debris is visible on the northern and eastern embankments. The eastern edge of the landfill drops four or five feet to a swampy area. Much of the debris consists of concrete, metal, wire, and other machinery-related items. A slight sheen was observable on the swamp water during one site inspection. Previous soil sampling by DOI on the north side of the field detected lead concentrations up to 553 mg/kg (Ruelle, February 1983). The landfill is suspected to contain ignitable magnesium metal, according to the Refuge Manager. An empty 30-gallon drum labelled "Magnesium Powder" was found along the southern end of the eastern face.

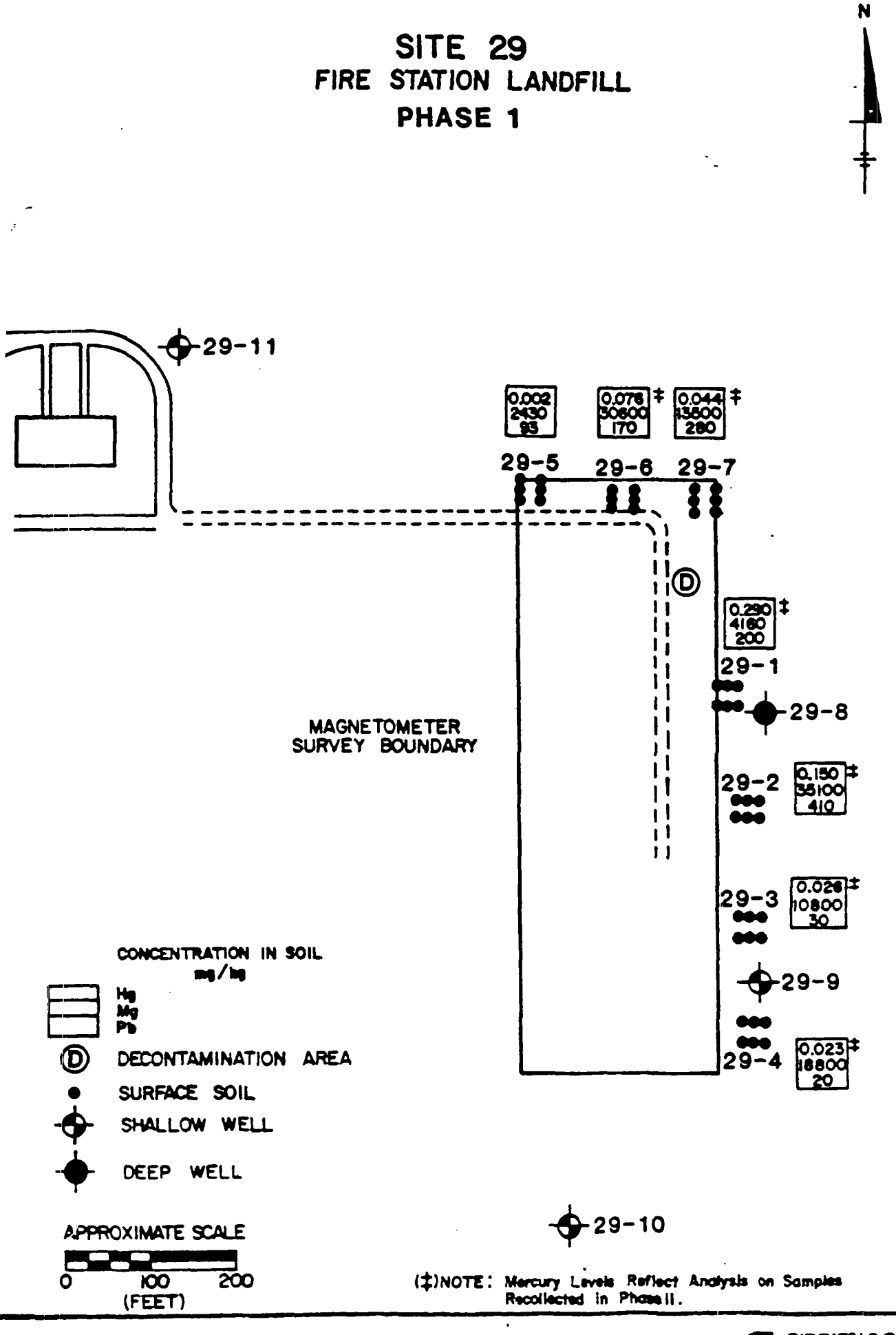
35.2 Site Investigations

35.2.1 Phase I Site Investigations:

A magnetometer and electromagnetic terrain conductivity survey was conducted over the northeastern end of the field using grid spacings of 20 feet. (See Figures 35-2 and 35-3).

Four ground water wells were installed. Three shallow wells were set within silty clay and silty sands to depths of 15, 25 and 30 feet. The fourth well was installed at a depth of 23 feet in sandstone bedrock. All wells screened the lower 10 feet of the boring.

SITE 29
FIRE STATION LANDFILL
PHASE 1



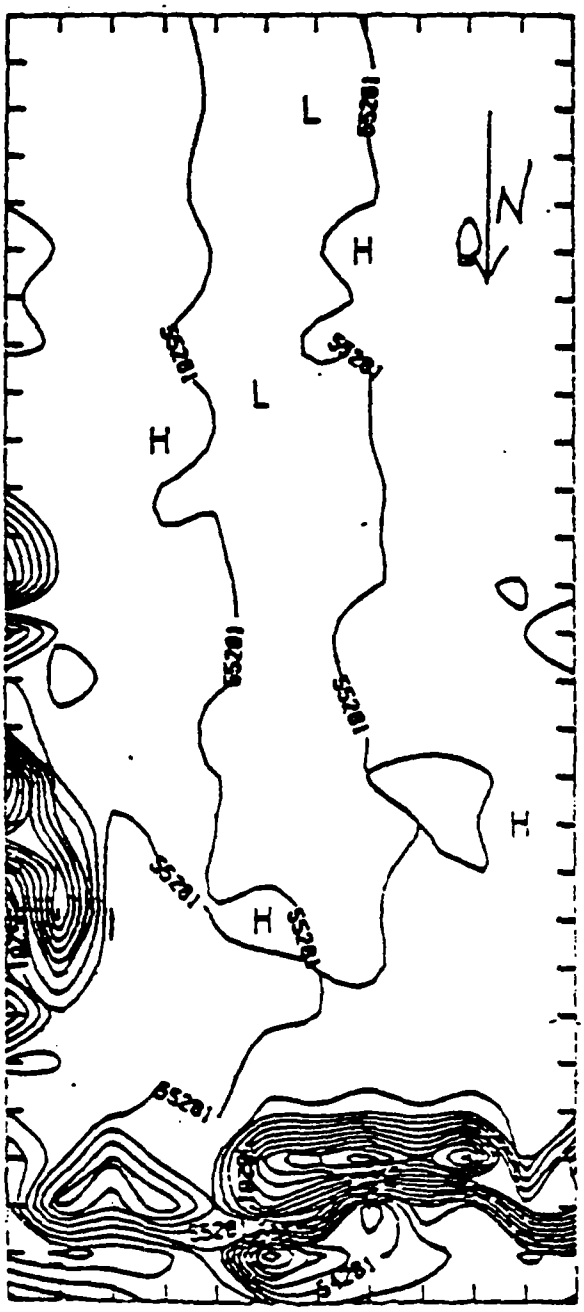
(‡)NOTE: Mercury Levels Reflect Analysis on Samples Recollected in Phase II.

O'BRIEN & GERE ENGINEERS INC

FIGURE 35-2

SITE 29

MAGNETOMETER SURVEY

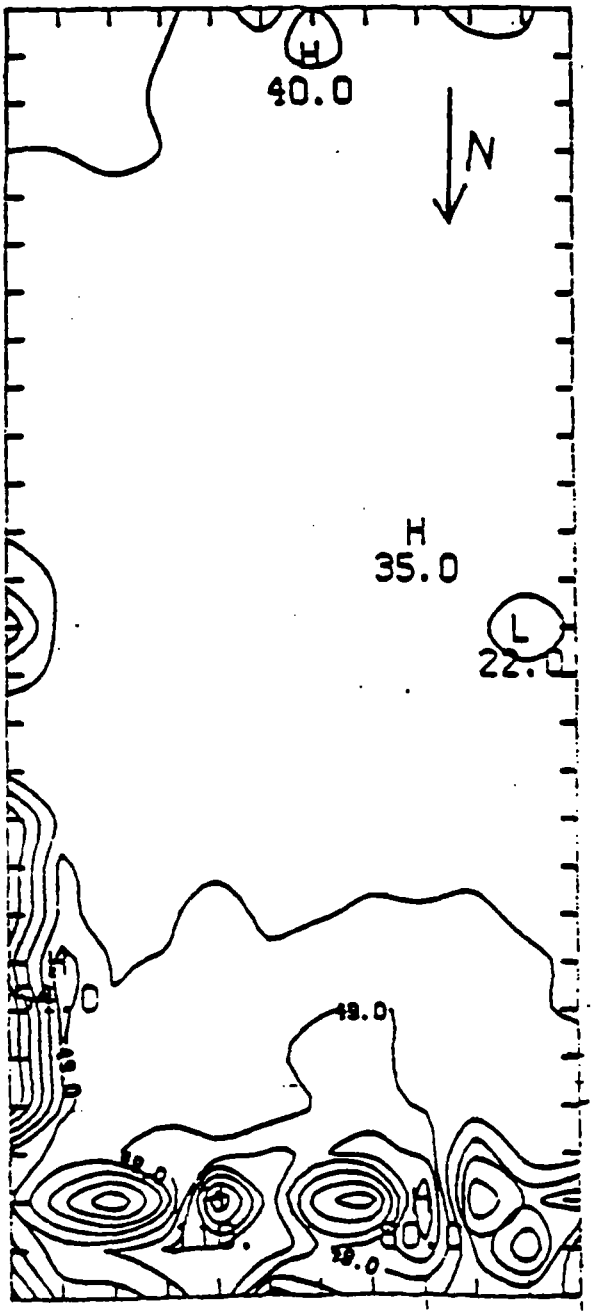


CONTOUR INTERVAL 5

O'BRIEN & GERE
ENGINEERS, INC

SITE 29

ELECTROMAGNETIC SURVEY



Seven composite soil samples (12 grabs each, at 0-1 ft depth) were collected at grid locations along the eastern and northern faces of the field. One soil from the eastern face was resampled for full organics analysis.

35.2.2 Phase II Site Investigations:

Ten test pits were dug to a depth of six feet, five pits along the east face and five along the north face. One composite soil sample was collected for each of the test pits along the east and north faces. Shallow (0-1 ft. depth) soil samples 29-1 to 29-7 were resampled at the Phase I locations for mercury analyses. Soil samples were also collected from the ditch approximately 50 feet from the landfill edge to prepare a field composite for the north and east ditch transects. The sample locations are shown on Figure 35-4. The soil samples were analyzed for PCBs, lead and magnesium.

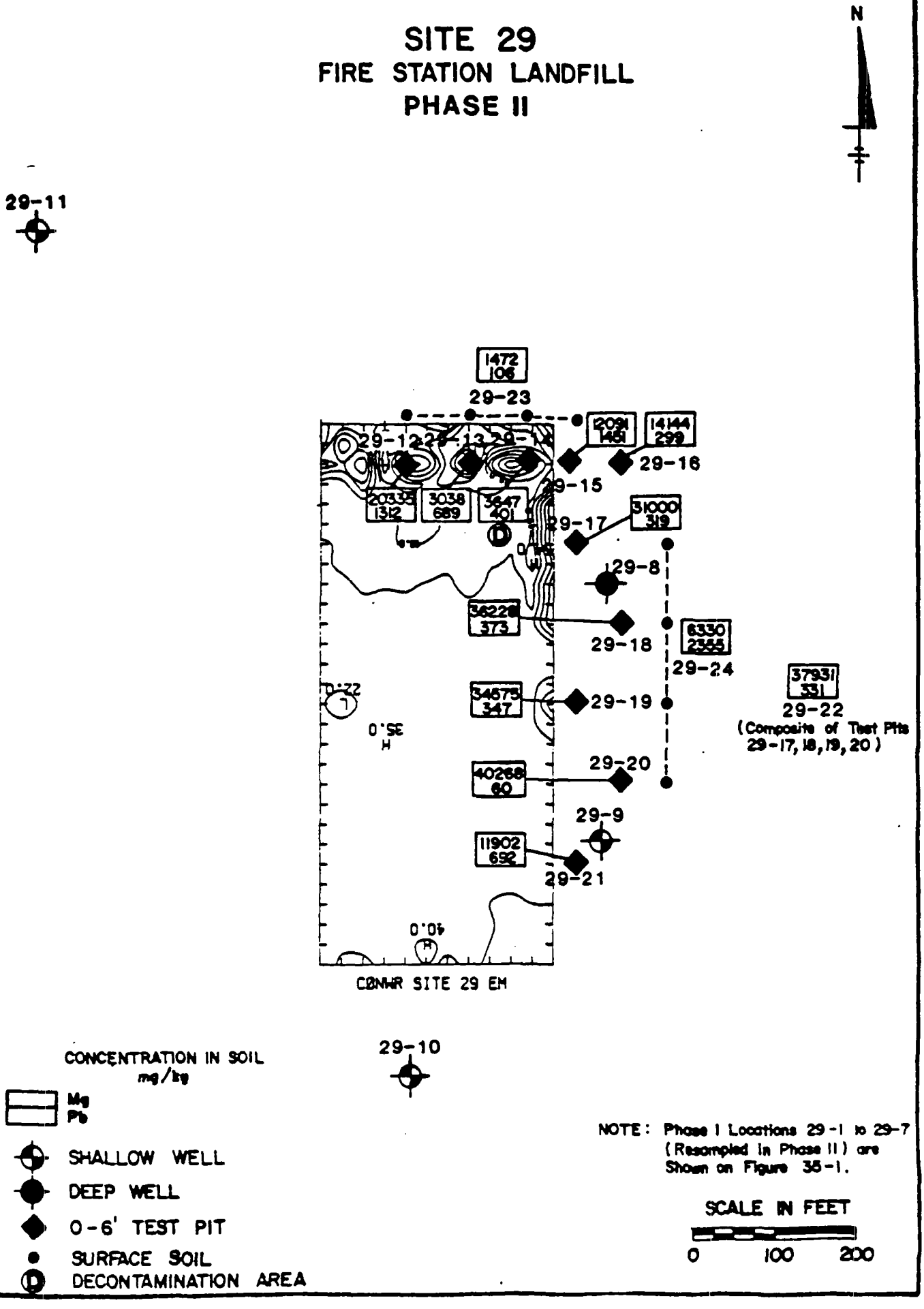
The four monitoring wells installed in Phase I were sampled and the ground water samples were analyzed for HSL CLP purgeables, pesticides, metals, and cyanide.

35.2.3 Site Hydrogeologic Characterization

35.2.3.1 Site Geology

Information obtained from the subsurface soil boring and well installation program (Borings 29-8 to 11) indicates that the site is underlain by approximately 13.5 to 28 ft. of unconsolidated sediments consisting of silty clay and clayey silt with trace sand content, overlying a light brown, medium grained sandstone bedrock. The bedrock encountered is similar to that encountered at most of the other sites. The

SITE 29
FIRE STATION LANDFILL
PHASE II



CONWR SITE 29 EM

silty clay/clayey silt overburden appears to be continuous throughout the site (see Appendix B). Bedrock is also laterally continuous, as it was encountered in all four on-site borings.

35.2.3.2 Site Hydrogeology

Occurrence of Ground Water

Shallow ground water occurring beneath the site was located consistently within the silty clay/clayey silt overburden. Based on inspection of saturation in soil core samples and ground water levels in wells, ground water in this unit appears to be unconfined similar to other areas investigated. Depth to ground water was found to range 1 to 7 ft. below ground level. Ground water elevations collected during the winter and summer of 1987 (wet and dry seasons, respectively) indicate a water table fluctuation of between 0.5 and 5 ft., with water levels dropping during the summer months (Table 4-3).

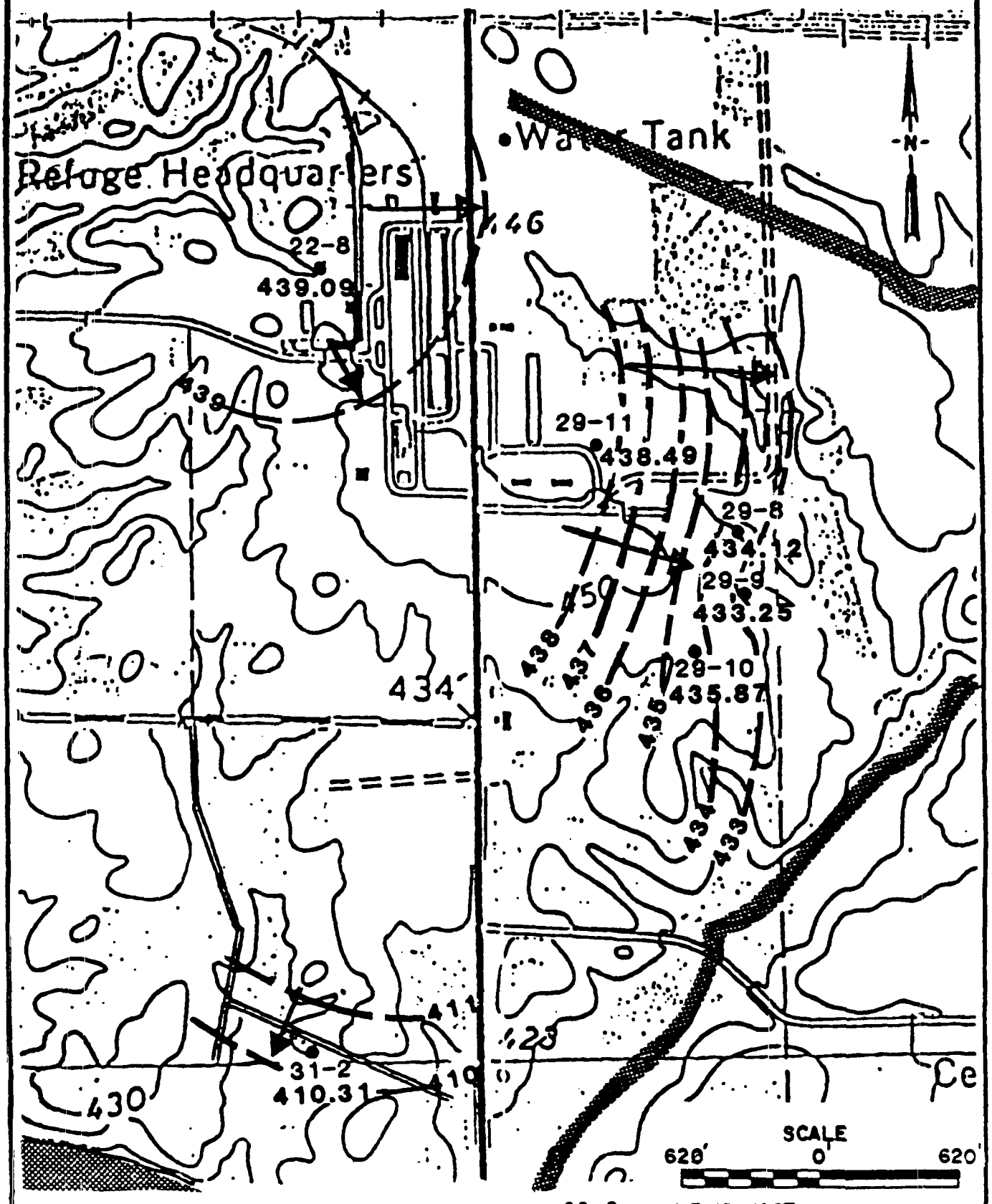
Ground Water Flow Conditions

Ground water elevations from the shallow ground water monitoring wells (29-8 to 11) were contoured and are presented on Figure 35-5. The ground water flow direction shown in this figure indicates a flow direction to the east, and towards an adjacent stream which flows southwest into Crab Orchard Lake. The average hydraulic conductivity (K) was calculated to be 1.56 ft/day. The hydraulic gradient (i) during June 18-19, 1987 was about 0.007 ft/ft as calculated from the four on-site monitoring wells. Porosity was estimated to be 0.35 (Davis and Dewiest).

A calculation was then made of the average ground water flow velocity (Vs) through the upper soil units. Using the formula given in

FIGURE 35-5

SITES 22, 29 &31 GROUNDWATER FLOW MAP



O'BRIEN & GERE ENGINEERS INC

Section 4.2, the resultant velocity was computed to be about 0.031 ft/day or 11.32 ft/year. The flow velocity is controlled by the relatively low hydraulic gradient and hydraulic conductivity of soils occurring in this area.

35.3 Analytical Results (See Appendix I, Page 28)

35.3.1 Phase I Analytical Results:

The magnetometer and electromagnetic survey shown in Figures 35-2 and 35-3 confirmed that the northern and eastern edges of the field are the areas containing metallic debris.

The metals values reported are estimated values to be used for screening purposes (see Exhibit B). Lead concentrations in the soil ranged from 20 to 280 mg/kg, consistent with the concentrations detected at the control sites, except for sample 29-1 from the east face which contained 410 mg/kg lead. Zinc levels of 23-929 mg/kg were approximately two times the zinc levels detected at the control sites. Magnesium was also somewhat elevated throughout the site ranging from 2,430 mg/kg to 35,100 mg/kg compared to levels of 1,210-1,380 mg/kg detected at the control sites. The highest lead concentration was found in the sample with the highest magnesium concentration. Mercury was detected in six soils along the east and north at concentrations ranging from below 1 ug/kg to 5.9 ug/kg but mercury analyses were questioned due to QA/QC deficiencies and thus this parameter was reanalyzed in Phase II. Sample 29-2 with the highest FID scan (19,123 ug/kg) of all the soils was analyzed for CLP organics, but none were detected.

35.3.2 Phase II Analytical Results:

The six soils resampled for mercury contained between 23-290 ug/kg, which are higher than background for the Refuge. PCB levels in the test pit soils ranged from less than 0.40 mg/kg to 2.10 mg/kg wet weight (less than 0.049 mg/kg to 2.56 mg/kg dry weight). Magnesium and lead were detected in all of the samples ranging from 1,472 mg/kg to 40,268 mg/kg and 60 mg/kg to 2,355 mg/kg respectively (see Figure 35-4). The samples collected from the landfill contained between 3,038 and 40,268 mg/kg magnesium, and between 60 and 1,451 mg/kg lead. The composite samples collected from the north and east transects of the ditch contained 1,472 and 6,330 mg/kg magnesium, and 106 and 2,355 mg/kg lead. The higher concentrations exceeded the levels for both metals at the control site soils by more than one order of magnitude.

The ground water samples contained acetone (23-11,500 ug/L), benzene (4 ug/L, below detection limit), iron (388 - 4,000 ug/L total and less than 25 ug/L dissolved), manganese (43 - 1,790 ug/L total and 24 - 1,770 ug/L dissolved), selenium (less than 2.5-41 ug/L total and less than 14-30 ug/L dissolved), and zinc (39-140 ug/L total and 16-78 ug/L dissolved). Cyanide concentrations were below the detection level of 0.05 mg/L. The parameters which exceeded the Illinois Public Water Supply and/or Federal Drinking Water standards were iron (standards of 1,000 (State) and 300 (Federal) ug/L), manganese (State and Federal standards of 50 ug/L) and selenium (State and Federal 10 ug/L); however, dissolved metals concentrations exceeded only the standard for manganese. Iron and manganese are regulated compounds due to their objectionable taste and color in water; the levels reported at this site do not pose a risk to humans or wildlife. One sample which contained 4 ug/L benzene was

above the AWQC for human health of 1.5 ug/L. Acetone was the only other organic detected but it was also present in the QA/QC blank.

Figure 35-1 illustrates the results for mercury, magnesium and lead at the site.

35.4 Environmental Effects

35.4.1 Qualitative Assessment

35.4.1.1 Source Evaluation

The Fire Station site is an abandoned field where various waste materials including mining machinery, are alleged to have been disposed of in the past. The area is currently well vegetated with brush and tall grasses. Some exposed waste materials are present along the north and east faces of the site. There is a low area adjacent to the landfilled area which contains standing water during wet weather months.

In general, little is known regarding the nature or origin of the waste materials. There were no containerized or free phase chemical wastes located during the sampling excavations. However, an empty drum with markings indicating that it had at one time contained magnesium metal was encountered. Based on the test pit explorations, it was determined that the site contained solid waste of mixed origin. The field team did not recognize any wastes such as beverage cans and food jars which might be classified as municipal refuse. There is no information or record available regarding the amount and chemical composition of the landfilled waste materials. Therefore, this risk assessment will be based upon the residues detected.

The results of the analytical investigations are consistent with the absence of chemical wastes at this site and the possibility that it may have been used for the disposal of munitions wastes. There were no organics detected in soils at this site. There were also no PCBs or pesticides detected. The only organic chemical detected was acetone, which was detected in ground water, but this may be attributed to residual acetone on the sampling equipment following decontamination with acetone. Acetone is also commonly detected as a lab blank contaminant. The chemical components detected, including zinc, lead and magnesium, are consistent with the potential use of the area for the disposal of munitions wastes. These metals, specifically magnesium and lead, are commonly encountered at munitions waste disposal areas. On the basis of the concentrations at which magnesium and lead were encountered and the potential acute and chronic toxicity of lead, magnesium and lead were chosen as site indicators for the risk assessment. Toxicological profiles for lead and magnesium are presented in Exhibit A.

35.4.1.2 Transport Route Evaluation

Air Route

The surface of the site is generally well vegetated. There were no large areas of exposed wastes prone to erosion and dust generation encountered during the course of the field investigation. Also, there was no evidence of vehicular or foot traffic over the waste site which might lead to the erosion of the cover and the generation of airborne dusts. Because there were no measurable residues of volatile organic materials detected in the soils sampled, it can be concluded that there are no volatile wastes present on the site surface which might evaporate and be

transported via the air route. On the basis of the above considerations, it can be concluded that the air route is not functional at this location. The air transport route will not be considered further in this evaluation except as applied to exposures to dust by burrowing rodents. Neither humans or non-burrowing wildlife would receive exposures to contaminated dust particles, discussed above.

Direct Contact

As stated above, the surface of the landfilled area is covered and generally well vegetated, with few exposed waste materials. Exposed wastes encountered are limited to a few pieces of scrap metal and debris. On this basis the direct contact route at the landfilled area is considered to be non-functional. However, the direct contact route will be considered in conjunction with the surface water transport route.

Ground Water

Analyses of samples of ground water collected from monitoring wells located at the periphery of the landfilled area contained detectable residues of acetone. It has been suggested in the source characterization section that the acetone residue may be an analytical or sampling artifact. However, this cannot be substantiated. Also, manganese concentrations above State and Federal standards were detected in ground water. Therefore, the residues will be accepted as reported. On this basis, the ground water transport route will be considered to be functional.

Surface Water

It has been established that the site is currently covered by a layer of vegetation and that there are few exposed waste materials in the landfilled area. Therefore, there is a low likelihood that waste materials are being eroded from the site by the action of surface precipitation. However, the landfill is situated adjacent to a steep grade which terminates at a ditch that drains into the wet area near the site. On the basis that a steep grade and a temporary surface water body exist near the site, it can be concluded that the surface water transport pathway can function at some time by scouring materials from the top of the landfilled areas and transporting them to the wet area. It is also possible that leachate may be generated and released along the face of the grade and be transported with runoff into the ditch and adjacent wet areas. These mechanisms could establish a source of waste materials for exposures via the ingestion and direct contact routes. On this basis, the surface water transport pathway will be concluded to be functional at this location.

35.4.1.3 Receptor Evaluation

The area where the landfilled waste materials are located is a considerable distance behind the Refuge Fire Station. This area is not used for recreational or industrial purposes. This area is also not near properties used for the propagation of agricultural commodities such as meat and dairy products or grain. The primary human activity nearest to this location is related to the Fire Station. Consequently, human activity in this area is expected to be very low. On the other hand, wildlife may inhabit the area. The low wet area and adjacent wooded areas may be

inhabited by wildlife such as small mammals and birds, as well as by invertebrates and amphibians. The area may also be the habitat for deer. There are no ground water users located in the area.

35.4.1.4 Summary of Complete Exposure Pathways

Based on the above considerations, it has been established that the air and direct contact routes at the landfill surface are incomplete, based on the non-functional nature of these transport routes. One eception is noted for the air route: exposure to contaminated dusts by small rodents which could burrow into landfilled areas. The ground water route has been determined to be functional based on the detection of ground water residues. However, there are no ground water users in the area. Therefore, the ground water route can also be concluded to be incomplete.

The primary route determined to be complete is the surface water route, because of the possibility for erosion and transport of wastes towards the adjacent ditch and wet areas, where wildlife or human recreational users may encounter the materials. This transport and exposure route will be considered in the quantitative assessment.

35.4.2 Quantitative Assessment

35.4.2.1 Estimate of Release and Exposure Rates

Surface Water Mediated Direct Contact Exposures:

The surface water transport route has been determined to be complete. Transport by this route could result in the establishment of residues of landfilled wastes in the ditch adjacent to the waste site and and other surfaces at downstream locations. A direct contact scenario involving wildlife or human receptors coming into contact with waste

residues in the ditch is possible, under a variety of circumstances. However, human exposures are much less likely to occur by this scenario than would be exposures experienced by wildlife inhabiting the wet areas.

Levels of lead detected in the landfilled areas ranged from 60 to 2,355 mg/kg. One composite sediment sample from the north ditch showed a lead concentration of 106 mg/kg, while a composite sediment from the east section of the ditch contained the highest lead level detected at 2,355 mg/kg. Assuming that humans will not repeatedly visit the site, exposures to soils in the ditch will be only on an acute (one time) basis. Previous evaluations presented in this report (see Section 24.4.2) assume that a human recreational user might ingest 100 mg soil during a four hour recreational visit to any particular site. If we assume exposures to soils in the ditch can occur highest concentrations of lead detected on the site, the following exposure can be calculated:

100 mg soil/visit x 2.35 ug lead/mg soil = 235.5 ug lead/visit,

235.5 ug lead/visit x 1/70 kg body wt. = 3.36 ug/kg/visit

The above calculation shows that, for a 70 kg human, this exposure represents a single dose of 0.00336 mg/kg/visit. Compared to a reported minimum chronic toxic dose of lead to a female subject over a 6 year duration of 450 mg/kg (RTECS, 1986) and a chronic no effect level of 0.32 mg/kg/day (USEPA HEA September, 1984), the dose calculated for this site represents a non-toxic level. For a worst case of three visits/year to this site, every year during a 70 year lifetime, each resulting in exposures of this magnitude, an annual lead exposure rate of 0.027 ug/kg/day is obtained, over 10,000-fold below the chronic no effect level. On this basis, it can be concluded that acute, and even chronic exposures potentially experienced by recreators who might spend a day in

the Fire Station Landfill area would not represent unacceptable exposures, even under the scenario of repeated exposures described above.

In contrast to the limited likelihood of human exposures at this site, wildlife, such as small mammals, birds and amphibians inhabiting the area could experience chronic exposures to residues of lead and magnesium present at the site. Since wildlife would likely roam over the landfilled areas as well as the ditches, it will be assumed that such species are exposed to lead levels equal to one-half the highest concentration detected at the site or 1,177 mg/kg lead. The typical wildlife direct contact exposure estimates used in previous sections have been based on a field mouse, deer, rabbit and bird; these species will also be applicable for the wildlife assessment at this site.

Using the soil ingestion and inhalation rates presented in the wildlife risk assessment for the Job Corps Landfill (Section 24.4.2), potential receptor wildlife species at this site may be exposed to the following levels of site lead residues:

		Estimated Daily (Chronic) Intake - Lead		
	Body Weight	Inhalation Rate	Ingestion	Total
Species	(kg)	ug/kg/day	ug/kg/day	mg/kg/day
Deer	60	NA	6.3	6.3
Mallard	1	[illegible]	[illegible]	11.2
Rabbit	1.0	NA	16.8	16.8
Mouse	0.03	2.3	11.2	11.2

NOTE: See Table 24-1 (Section 24.4.2.2) for assumptions.
NA = Not Applicable

t oug t e e a e o i e t o i toxicity data on these species of wildlife, these dosages of lead, when evaluated in comparison to the previously cited USEPA chronic no-effect level of 0.32 mg/kg-day for humans (which was generated on the basis of rat studies) indicates that

the doses calculated above may represent levels at which chronic toxicity could be experienced by wildlife.

35.4.2.2 Toxicological Assessment

The above analysis indicates a low likelihood for humans to come into contact with wastes present in the vicinity of the landfilled materials. However, even in the case where a human might encounter and ingest "worst case" amounts of eroded wastes, the dose experienced would not be likely to cause the human to experience toxic effects.

In contrast to the limited potential for the waste materials to represent an unacceptable risk to human receptors, it is possible for wildlife in the area to ingest residues of lead associated with eroded wastes at concentrations which would represent toxic effects under a chronic exposure scenario.

35.4.3 Analysis of Uncertainties

The following are the major factors which should be con- sidered when evaluating the uncertainties associated with this evaluation:

1) The waste materials present within the landfilled areas have not been comprehensively characterized. Therefore, the actual amounts, physical state and chemical composition of the waste source may not be consistent with the identities and concentrations utilized in this evaluation.

2) There is limited information available on the concentrations and distribution of waste materials present in the ditch and wet areas evaluated as part of the surface water mediated direct contact

exposure route. The concentrations assumed in the quantitative assessment may be overestimated, thereby overestimating the actual risks experienced by wildlife.

3) There are no data available on the chronic toxicity of lead to wildlife suitable for use in evaluating the residues predicted at the site studied. The chronic effect level used in this assessment to evaluate potential toxicity to wildlife has been derived for humans, on the basis of laboratory animal studies. Differences in body weight and surface area, as well as metabolic considerations and wildlife disease states, represent an unquantifiable level of uncertainty which could effect the reliability of the risk estimates regarding wildlife presented in this assessment.

35.5 Preliminary Remedial Alternatives

Based on the site investigations and the analyses of environmental effects, remedial actions aimed at reducing exposure to lead residues in soil by wildlife might be required. In addition, removal or containment of soil may be required in some areas of the landfill, particularly if additional characterization shows the presence of EP Toxic Sediments.

The contaminants detected above Refuge background levels include lead, magnesium, mercury and zinc in soil matrices. Most soil samples including surface and test pits (to a depth of 6 ft.) were taken along the limits of the fill area, as evidenced by the grade along the north and eastern edges, and thus exhibited high metals concentrations. A field transect taken approximately 50 ft. away from the grade apparently was beyond the limits of the fill along the north face, since it contained levels of magnesium and lead similar to the

range for Refuge background soils. The limits of the landfill along the eastern face may extend beyond 50 ft. from the eastern grade since the data for the east face transect sample contained levels of lead and magnesium similar to the levels found in the samples from the edge of the landfill.

The unfiltered samples of ground water contained iron, manganese, and selenium above the State Public Water Supply standards, but the metals were mostly associated with particulates and thus would not be expected to migrate readily in the groundwater table. Moreover, iron and manganese constituents were not considered to pose any health concerns at the levels detected.

Any future land uses should be evaluated prior to approval by the Refuge Management. Some of the potentially applicable remedial actions for this site are discussed below. A summary of the potential response options for this site is included in the Executive Summary (Table 2).

Limited Site Access

It may be advisable to limit human and wildlife exposure to water or soils at the Fire Station Landfill through construction of a fence around the site. Access could be controlled by the installation of locked gates to ensure that only authorized personnel enter the site. Deed restrictions might be imposed to limit future uses of the site.

Removal or Containment of Soil

One alternative to control the migration of metals offsite might involve capping the edges of the landfill with a clean, relatively impermeable cap. Additional sampling may be warranted to further define

the limits of the landfilled area and the extent of contamination in the east wetlands. EP Toxicity analyses might be conducted to determine if the lead wastes are hazardous.

Monitoring - Ground Water

Follow-up remediation efforts may include periodic sampling and analyses of the four monitoring wells for lead, magnesium, manganese, mercury, and zinc.

In addition, studies to verify the adequacy of the cleanup might begin immediately following remediation.

35.6 Conclusions and Recommendations

It can be concluded that the Fire Station Landfill site is impacted with the principal pollutant being lead. It is recommended that remedial alternatives for this site be evaluated in the FS. Potentially applicable remedial measures include removal or containment of contaminated hot spots, surface water control measures and ground water monitoring.

SECTION 36 - SITE 32, AREA 9 LANDFILL

36.1 Site Description

Area 9 is a manufacturing site on the Refuge. It was leased to Sangamo Electric Co., Capacitor Division from 1946 to 1962 and is currently leased to Olin Corporation. The area is comprised of a Building Complex and an inactive landfill located adjacent to the buildings (See Figure 36-1). Sangamo Electric manufactured various types of capacitors, utilizing aluminum, electrolytes, mica, silver, lead foil, and PCBs. Olin Corporation currently uses the buildings to manufacture explosives. Over the years, a number of other companies have occupied Area 9, according to the Refuge files, including, but not limited to, machining and metal fabricating industries, electrical components, and explosives manufacturing industries.

The Area 9 Landfill was reportedly used during the 1950s and early 1960s for disposal of wastes from capacitor manufacturing operations. The landfill was closed in 1964 (Ruelle, July 1984). When the landfill was active, wastes were burned, compacted in a swale and covered (Adams, May 24, 1984). Specific types of components that may have been disposed include some containing lead, acetate, PCBs (Aroclor 1254 and 1242), and PCB combustion products. Other possible disposed materials from capacitor manufacturing may have contained mica, silver, cyanide, aluminum hydroxide, aluminum oxide, gold, copper, zinc, hydrochloric acid, styrene, nitric acid, phosphoric acid, and borates. Other industrial wastes may include cyanides, printing inks and lead-based explosives. In addition, waste oils and debris were reportedly burned and covered with soil. The landfill was also possibly used for disposal of wastes generated in the manufacture of explosives. The landfill is locatec

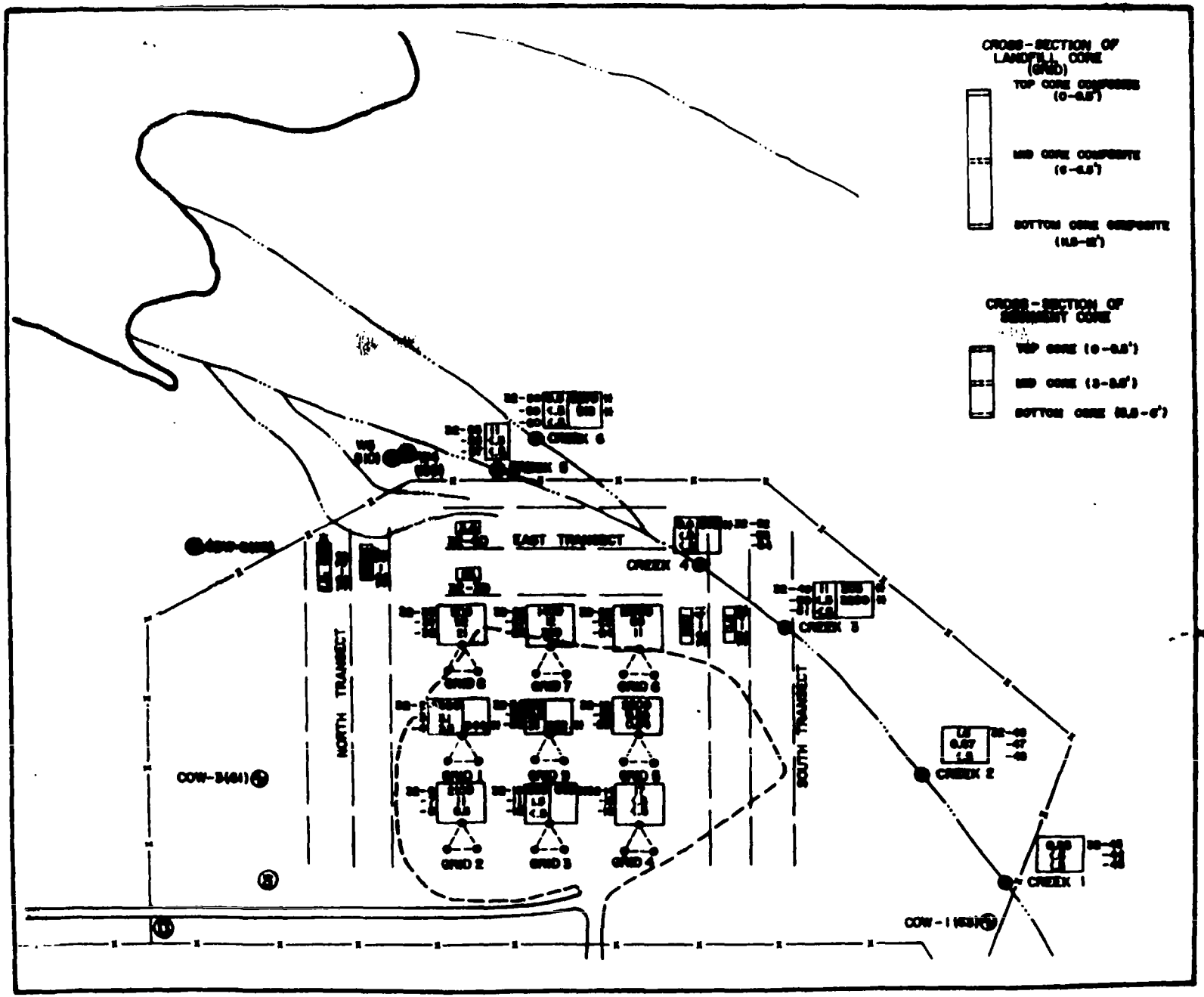


FIGURE 32-1

SITE 32
AREA 9 LANDFILL

PHASE I

LEGEND

LANDFILL BORING GRID LOCATION

SEDIMENT BORING LOCATION

PCB CONCENTRATION (mg/kg WE
LEAD CONCENTRATION (mg/kg DR
PCB LEAD (Phase II Results)
TOP CORE
MID CORE
BOTTOM CORE

LOCATION WHERE LE CONCENTRATION EXC BACKGROUND CONCENT AT CONTROL SITES.

DECONTAMINATION AR

WASTE STORAGE AI

NOTE: See Appendix F for Sampling I Locations.

APPROXIMATE SCALE IN FE

0 80

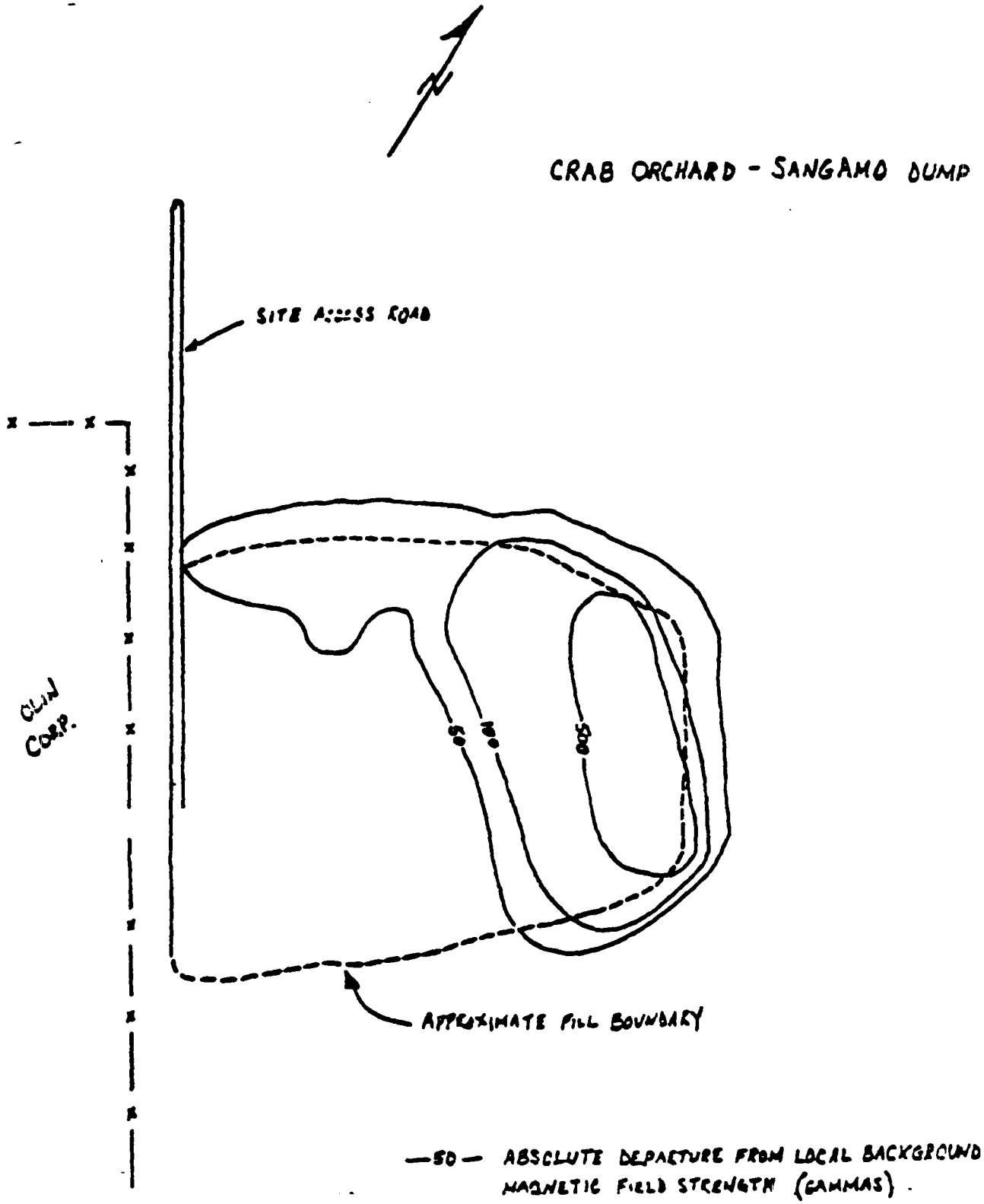
about 100 yards south of Crab Orchard Lake and 100 yards east of the building complex.

The limits of the landfill are discernible by changes in the topography and vegetation, revealing an area of approximately 2.5 acres with an estimated fill thickness of 8 to 10 feet in the middle and 6 feet at the edges. The landfill is covered by a thick growth of tall grass except where waste materials are exposed. The volume of the landfill is estimated to be from 16,000 to 35,000 cubic yards. Materials visible on the surface appear to be electrical components consisting of small capacitors, capacitor parts, chunks of a golden resin, and a number of 3-inch steel cuplike pieces.

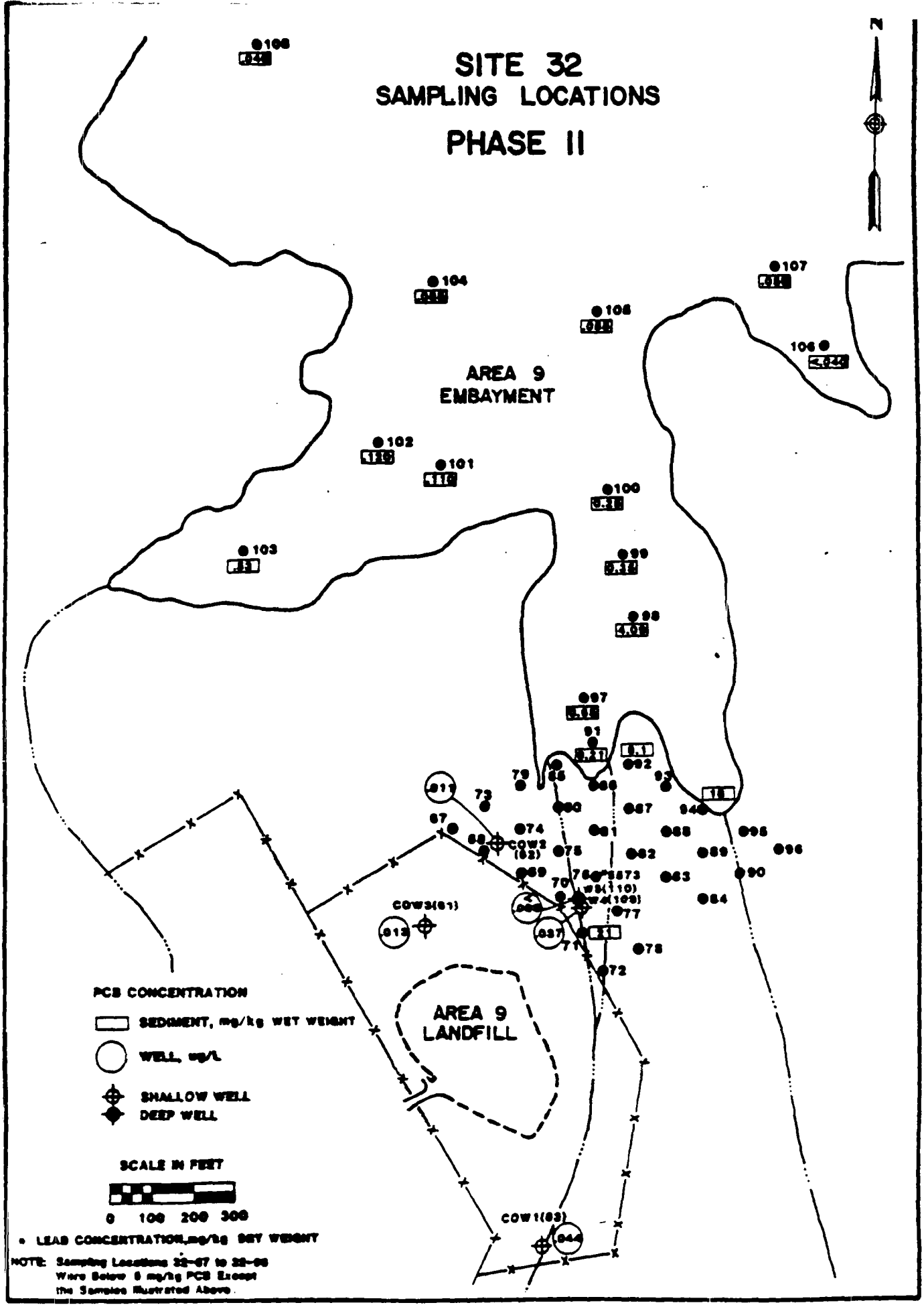
A magnetometer survey conducted by DOI suggested that the majority of wastes are buried along the eastern and northern edges of the landfill. (See Figure 36-2). Runoff from the landfill can drain into an intermittent creek and into Crab Orchard Lake.

Three ground water monitoring wells were installed in the vicinity of the landfill in late 1983 prior to the Phase I investigation (Gifford, 1984). (See Figure 36-1, Wells COW1-COW3). The wells were installed in silty sand to depths of 29, 35.5, and 30.5 ft., and screened the bottom 5-10 ft. of the boring. Boring logs from previous investigations by Illinois EPA at the Area 9 landfill reveal that the site is underlain by a minimum of 7.5 feet of clay. The silty clay is reported to be upwards of 30 feet thick north of the landfill and 25 feet thick south of the site. Groundwater, at locations around the landfill, was reported to be 2 to 24 feet below the surface during different time periods (Gifford, 1984).

SITE 32
MAGNETOMETER SURVEY



SOURCE:
Byram, Scott. Memo to File. Crab Orchard National Wildlife Refuge– Sangamo Dump. TDD No. R5-8308-6. September 21,1983.



O'BRIEN & GERE
ENGINEERS INC.

36.2 Site Investigations

36.2.1 Phase I Site Investigations:

Nine compositing stations were established within the landfill area for collection of soil samples. (See Figure 36-1). Samples were obtained from one-foot intervals from the surface to a depth of 12 feet.

The exact boundaries of the landfill are unknown because contaminants could have washed from elevated portions of the landfill onto the lower surrounding area. To identify the extent of contaminant transport from the landfill to surrounding areas, surface soil subsamples were collected at 3-foot intervals along each of six transect lines, two each on the east, south and west side of the landfill as shown on Figure 36-1. The soil composites at each depth (top, mid, and bottom cores) were analyzed for PCBs, indicator compounds and PCDD/PCDF screening. A composite for each grid location (0-12 ft depth) was analyzed for the full priority pollutants screen.

Grab sediment samples (0-1 ft depth) were also collected from six locations along the intermittent creek adjacent to the landfill. The sediments were screened for priority pollutants, metals, explosives, and cyanide.

36.2.2 Phase II Site Investigations:

Thirty surface soil samples were collected from the lowland area northeast of the landfill. These samples were collected from a 5 sample by 6 sample grid with grid points approximately 100 ft. apart. Twelve sediment samples were also collected from the lake embayment downstream (north) of the landfill. (See Figure 36-3). The soil and sediment samples were analyzed for PCBs and lead. Five of these soils were analyzed for full CLP organics also.

Twenty-four surface and nine core soils collected in Phase I were reanalyzed in Phase II for mercury, chromium, and lead. Mercury was reanalyzed because the Phase I results were questionable due to QA/QC deficiencies.

Three additional monitoring wells were installed during Phase II (Figure 36-3). Well 32-63 was installed south of the landfill near well COW-1, one of the three wells installed in 1983. COW-1 could not be located. Further information describing the wells installed prior to the RI/FS is provided in Section 36-1. Wells 32-63 and 32-109 in Phase II were installed in silty and silty clay soils to a depth of 15 feet and screened from 10 to 15 feet. Well 32-109 was nested with well 32-110. Well 32-110, an artesian well, was installed on top of bedrock at 82.5 feet and screened within sandy soils from 77.5 to 82.5 feet. The five monitoring wells (including the wells installed prior to this RI) were sampled and ground water samples were analyzed for full CLP organics, metals, low level nitrosamines and cyanide.

In addition to the soil and water samples, several capacitor parts were found either buried or partially buried along the drainage ditch northeast of the Building Complex. The capacitor casings and contents were analyzed for PCBs and lead, as were several soil samples collected from the area where they were found.

36.2.3 Site Hydrogeological Characterization

36.2.3.1 Site Geology

Information obtained from the subsurface soil boring and well installation program indicate that the site is underlain by approximately 77 feet of unconsolidated sediments overlying sandstone bedrock. The

particular sequence of soil units encountered in the deep well No. 32-110 consisted of alternating coarse to fine sand and silty clay units 5-25 feet thick. Similar to other sites, the bedrock was overlain by a sand layer. This area contained, however, additional sand layers not found in other sites. Since no other deep borings were drilled in this area, the lateral extent of the sand layers is not known. Typical of the area, the bedrock encountered at a depth of 77.5 feet consisted of a gray, medium grained, sandstone with the upper 1-2 feet highly weathered.

36.2.3.2 Site Hydrogeology

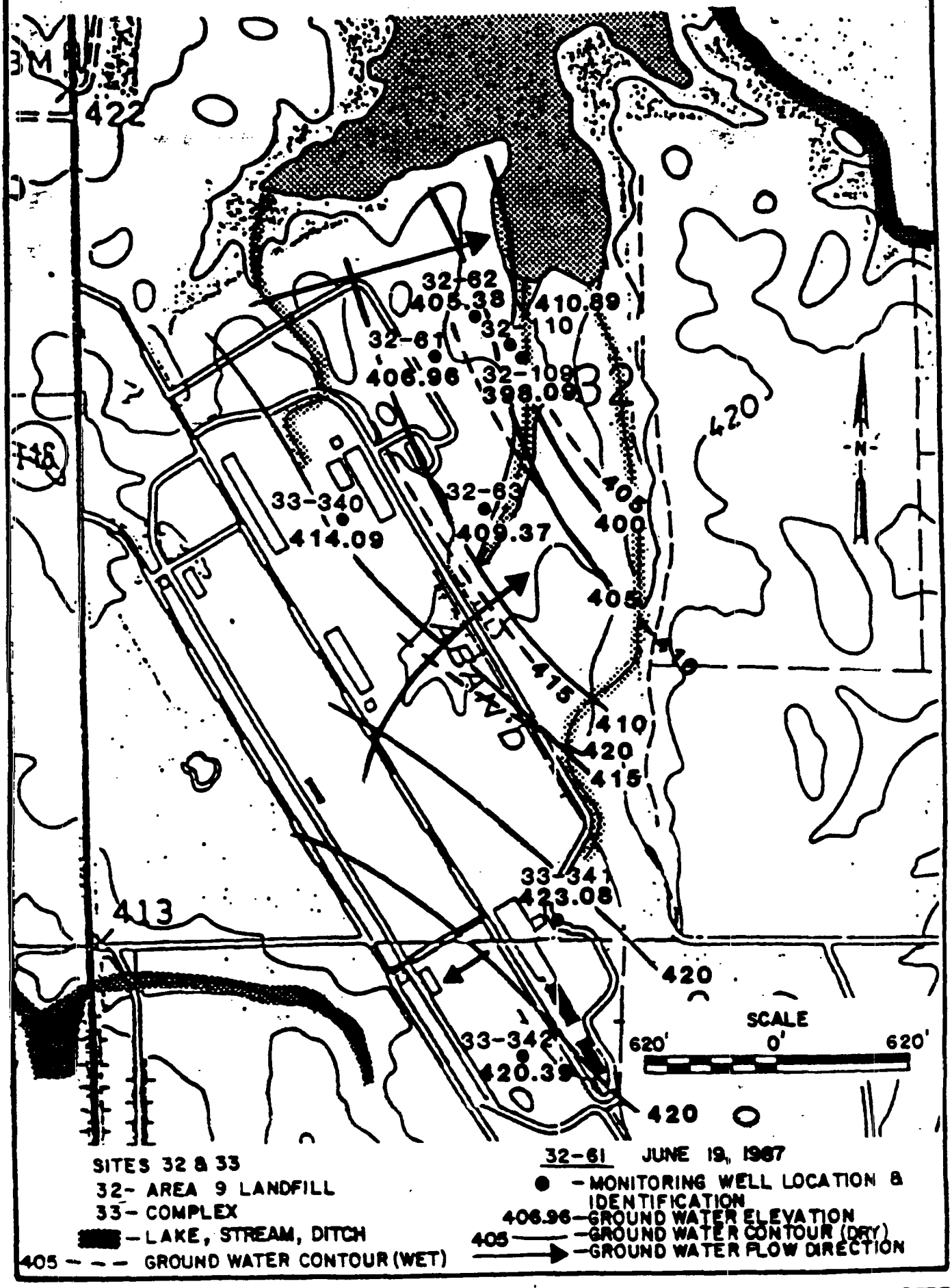
Occurrence of Ground Water

Shallow ground water occurring beneath the site was located about 5-12 feet below ground surface in June 1987 within upper silty clay and fine sand layers. Ground water levels were higher during the wet season (Dec. 1986) by about 3-10 feet. Ground water in these units appears to be unconfined similar to other areas investigated. Ground water occurring within lower portions of the soil sequence on top of bedrock was found to be confined and yielded a continuous flow at ground surface of about 5 gal/min. Ground water within underlying bedrock was not investigated.

Ground Water Flow Conditions

Ground water elevations from the shallow monitoring wells were contoured and are presented on Figure 36-4. The ground water flow direction exhibited in this figure indicates a flow to the northeast toward Crab Orchard Lake. The hydraulic gradient of flow (i) during June 18-19, 1987 was about 0.015 ft/ft. The average hydraulic conductivity

SITES 32 & 33 GROUNDWATER FLOW MAP



O'BRIEN & GERE ENGINEERS INC.

(k) was calculated to be about 0.56 ft/day from the five shallow monitoring wells. The porosity (n) was estimated to be 0.35 (Davis and Dewiest).

A calculation was then made of the average ground water flow velocity (Vs) through the upper soil units. Using the formula given in Section 4.2, the resultant velocity was calculated to be about 0.024 feet/day or 8.7 feet/year. The flow velocity is controlled by the relatively low hydraulic gradient and hydraulic conductivity occurring in this area.

An upward vertical flow potential was identified from the unconsolidated aquifer screened by the deep well 32-110 into the upper water table. This phenomenon indicates discharge of deep ground water toward adjacent Crab Orchard Lake, similar to the shallow ground water.

36.3 Analytical Results (See Appendix I, Page 29)

36.3.1 Phase I Analytical Results:

Figure 36-1 presents the results of PCB analyses within the Area 9 landfill. The PCBs are generally present above the TSCA criterion of 50 mg/kg only on the surface (0-1 ft) except for the eastern edge of the landfill area. PCBs above 50 mg/kg were observed at the 6 ft depth in the northeast (grids no. 6 and 8). Grid number 7 was the only station where soils contained PCBs above 50 mg/kg at the 12 ft depth. (See Figure 36-1).

PCB levels in the soil samples from the transects on the north, south and east sides of the fill area were above 50 mg/kg at the surface, immediately adjacent to the landfill but were below 50 mg/kg on the adjacent transect on all sides. (See Figure 36-1). Sediments in the

drainage channel upstream and downstream of the landfill contained wet weight PCB concentrations of 0.86-1.6 mg/kg, and 0.8-11 mg/kg, respectively, while PCBs were undetected (less than 0.5 mg/kg) for all the 3 ft. and 6 ft. depths of the downstream sediments.

Lead concentrations ranged from 11 to 29 mg/kg in 0-3 ft. composite sediments from the creeks around the landfill, and from 0.9 to 25 mg/kg in the 0-12 ft. composite soil samples from the landfill that were analyzed for lead. However, the metals concentrations reported are not supported by QA/QC (see Exhibit B). Selected Phase I grab soil samples which had not been previously analyzed for lead were resubmitted during the Phase II program for analysis of lead, mercury, and chromium. The reanalyses were considered necessary due to the uncertainty inherent in the Phase I analyses, and also to better quantify the levels present in the grabs at specific depths. Nine out of 32 soil samples reanalyzed showed lead concentrations outside the common range of 2-200 mg/kg in soils (Lindsay, 1979). Figure 36-1 shows the lead concentrations for these 9 samples ranging from 205 to 8,570 mg/kg.

The remaining samples showed lead levels similar to the Refuge control sites. Figure 36-1 shows the results for lead from the Phase II reanalyses. Further discussions of the metals analyses are given in the next section, Phase II Analytical Results. Full analyses of organic priority pollutants were conducted on each of the nine boring composites (0-12 ft depth). The concentrations of heavy metals and other contaminants present in the composite samples from the landfill did not differ significantly from those at the control sites.

The results for PCDD and PCDF isomers in soils are presented on page 30A of Appendix I. A separate data listing is included to illustrate

the actual dioxin/furan to PCB ratios compared to the ratios which would be expected based on the PCB concentration for that sample. The purpose of calculating this ratio is to determine if dioxins/dibenzofurans are elevated due to burning products of PCBs. If this were the case, then the dioxin/furan to PCB ratios would be higher than the expected normal ratios for PCBs. The Table lists a concentration for each peak detected in the scan (Isomer No.) and a total concentration for each compound. This total concentration was compared to the average PCB concentration detected in the same sample to develop a ratio of PCDD or PCDF to PCBs. An average PCB concentrations was used because the analyses were conducted in duplicate (one reported by ETC Laboratories and one reported by OBC Laboratories).

Based on studies conducted by T. Sawyer and S. Safe et.al.(1982, 1984, 1985) relative to PCB isomers and congeners, the expected fraction of PCDD and PCDF isomers associated with a measured PCB concentration can be calculated. The 'equivalent' fraction of dioxins and furans is determined by a conversion factor or Keq value. The Keq value is based on 1.00E-05 for 2,3,7,8-TCDD.

The results for Area 9 Landfill showed positive detections above the ratios which would be expected for TCDF, PCDF, and OCDD compounds. TCDF levels ranged from 0.14 to 26.3 ug/kg in 7 out of 12 samples. PCDF was detected in one sample at 0.34 ug/kg. OCDD levels ranged between 0.6 and 20.6 ug/kg in 9 samples. There was no specific pattern discernable in the distribution between dioxins and furans.

36.3.2 Phase II Analytical Results

PCB concentrations in surface samples from the lowland area downgradient of the landfill were all below 5 mg/kg with the exception of three of the thirty samples taken. These excursions were noted at Locations 32-71, 32-92, and 32-94, with 21,8.1, and 18 mg/kg wet weight, respectively. (See Figure 36-3).

PCB concentrations in sediment samples from the lake embayment area near the landfill showed a maximum concentration of 4.09 mg/kg, while all other lake sediments (locations 32-97 through 32-108) contained PCB concentrations below 0.65 mg/kg. (See Figure 36-3).

The five ground water monitoring wells sampled during Phase II all exhibited PCB concentrations below 0.045 ug/L. All downgradient ground water concentrations (less than 0.005 to 0.037 ug/L) were less than the upgradient concentration of 0.044 ug/L. There are no state or federal standards for PCBs in water. However, PCBs in four of five monitoring wells exceeded the Ambient Water Quality Criteria for human health. PCB concentrations detected in ground water are likely associated with suspended particulates in the well, since these compounds have a high affinity for and adsorb tightly to the silty clay soils at the site. The presence of PCBs in suspended solids in the wells may result from contaminated surface sediment introduced during installation of the well, natural fluctuations in water levels causing particulates from the upper soil layers to be washed to the lower levels where the well is screened, or by adsorption of soluble PCBs onto the suspended matter already present in the well. Well 32-63 was found to contain 92 mg/L of chromium, which exceeds the Illinois Public Water Supply Standard and Federal MCL of 50 mg/L. However, the dissolved chromium concentration of 1.2 ug/L for this sample was within all applicable standards.

Acetone and methylene chloride were detected but were also present as contaminants in the QA/QC blanks. All other concentrations were within these standards. Of the 30 Phase II soil samples from the low lying area northeast of the landfill that were analyzed for lead, only one sample (32-76) contained lead concentrations significantly in excess of the concentrations detected at the control sites, with 5573 mg/kg (See Location 32-76, Figure 36-3). All other concentrations were well below the background level of 200 mg/kg. The Phase I soils from the landfill reanalyzed under Phase II generally contained concentrations within the refuge background levels for chromium.

Mercury levels ranged from less than 0.023 to 0.035 mg/kg, which are similar to the concentrations found at the control sites. Lead concentrations in the landfill were above the typical range of 2-200 ppm in soils (Lindsay, 1979) for nine out of 32 grab soils that were reanalyzed in Phase II. The lead results for selected samples which ranged from 205 to 8,270 mg/kg are shown in Figure 36-1. The lead concentrations in the remaining samples were similar to the levels detected at the control sites. Only traces of organic compounds other than PCBs were detected at the landfill.

Acetone, methylene chloride, di-n-buthylphthalate and isophorone were detected but these were also present in the QA/QC blanks. The analytical results for the capacitors found at Area 9 showed positive detections for PCB Aroclor 1254 for the cylindrical and square capacitors. (See Page 30B of Appendix I). A composite of five smaller capacitors tested positive for PCB Aroclor 1242. Some capacitors did not contain detectable concentrations of PCBs. The soil samples collected from the area where the capacitors were found also contained PCB Aroclor 1254 at

concentrations between 1,100 and 88,000 mg/kg wet weight. The capacitor casings were also analyzed for lead and were found to contain between 156 and 20,000 mg/kg wet weight of lead. The soil samples contained lead ranging from 1,810 to 20,500 mg/kg wet weight (1,880 to 37,600 mg/kg dry weight).

36.4 Environmental Effects

36.4.1 Qualitative Assessment

36.4.1.1 Source Evaluation

The results of the site investigations, as described in the preceding sections, determined that the Area 9 Landfill was used for disposal of wastes arising from the manufacture of capacitors. A variety of waste materials were disposed of at the site, including lead and PCBs. Most of the wastes are situated on the western and northern boundaries of the landfill. Runoff from the landfill can reach Crab Orchard Lake via intermittent creeks. The results of the Phase I and Phase II investigations indicate that PCBs are the major contaminants identified within the landfill site itself, while lead was the principal contaminant found in the lowland area below the landfill.

PCB levels were generally highest in the northern and western sectors of the landfill, with values in the 0 to 1 foot cores ranging from 2,100 mg/kg up to 13,000 mg/kg (Figure 36-1). Subsurface contaminant levels were considerably lower. Sediments taken from intermittent creeks draining the landfill showed low-level PCB contamination (0.8 to 11 mg/kg in six creeks sampled). Several soil samples from the landfill and some off-site creek sediments reanalyzed in Phase II contained elevated

levels of lead, ranging from 205 to 8,270 mg/kg, although no pattern of contamination was evident.

Isomers of tetrachlorodibenzofurans (TCDF) ranging from 12 to 28 ug/kg were detected in three soil samples at this site. Detection of polychlorinated dibenzo/dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) present a cause for concern due to the wide range of acute and chronic toxicity observed for one member of these classes; 2,3,7,8-tetrachloro- dibenzodioxin (2,3,7,8-TCDD). 2,3,7,8 - TCDD is highly toxic under acute exposure, and produces a number of chronic disorders including but not limited to immunotoxicity, teraogenicity and reproductive toxicity, and it is a suspect human carcinogen. There are very strict molecular requirements for production of toxic effects by these chemicals at a biochemical level, due to required interactions with receptor molecules in target tissues. 2,3,7,8-TCDD, and to a lesser extent 2,3,7,8-TCDF appear to have the optimum spatial and electronic requirements for toxic potential of all the isomers in these two classes of chemicals. However, there is no evidence at present that 2,3,7,8-TCDF or other isomers are carcinogenic in either animals or humans. Removal of chlorine or substitution at other ring positions greatly diminishes observed acute and chronic toxic effects relative to 2,3,7,8-TCDD. Addition of chlorine to PCDDs and PCDFs also decreases toxicity. Thus, a mixture of hexachlorodibenzodiozins was shown to be only a weak liver carcinogen in rodents while octachloro-dibenzodiozin is non-carcinogenic. In view of detection of a TCDD in only one site sample, lack of information on the carcinogenicity of the TCDFs and on the isomeric composition of the site TCDI and TCDF, quantitative risk estimates for these site contaminants are inappropriate at present.

Chromium was found in one groundwater sample in excess of Illinois Public Water Supply Standards and the Federal MCL. However, the dissolved chromium concentration for this sample was within all applicable standards. Trace PCB concentrations below 0.045 ug/L were detected in the ground water samples. Based on this analytical survey and the non-threshold and threshold toxicity of these chemicals, PCBs and lead were chosen to serve as site-indicator contaminants for the purpose of this risk assessment. The physicochemical and toxicological properties of PCBs and lead are summarized in Exhibit A. Both PCBs and lead adsorb tightly to the silty and silty clay soils observed at this site (see Section 36.2). This property is the dominant factor determining the environmental transport and fate of these compounds. The measured soil hydraulic conductivity at this site ranges from 4.0×10^{-6} to 2.32×10^{-6} ft/sec. The transport of PCBs and lead are significantly retarded by this type of soil.

36.4.1.2 Transport Route Evaluation

a) Air: Exposures to PCBs in the vapor state as a result of volatilization from high residue levels in soil were discussed in Section 24.4.2.1, and were determined to contribute approximately 8 percent of the total airborne exposure. In addition, the existence of exposed soil containing tightly adsorbed contaminants in some areas of the site, dusts generated by wind erosion, or the activities of endemic wildlife constitute a functional route for conveying PCBs and lead to on/off-site locations via the air route for subsequent exposures by receptors in those areas.

b) Direct Contact: Due to the presence of site indicator contaminants in exposed wastes, soils, and sediments in the area, exposures by the direct contact route are possible.

c) Surface Water: One main creek northeast of the landfill serves to drain the site and Crab Orchard Lake is 100 yards downgradient of the landfill. Furthermore, sediment analyses from the creeks showed the presence of both PCBs and lead. Therefore, the surface water transport route is considered functional via precipitation-initiated runoff events which convey soil- and sediment-bound site contaminants towards Crab Orchard Lake.

d) Ground Water: Estimates of contaminant transport via groundwater were prepared for PCBs and lead. Using the groundwater flow velocity calculated in Section 36.2.3 (0.024 ft/day) and an average PCB concentration of 0.020 ug/L for the three shallow downgradient monitoring wells, a worst case estimate of 0.20 mg/day PCBs could potentially reach the lake. This is based on a plume depth of 30 ft and width of 500 ft, since, based on the analysis of one deep well (32-110) which was screened at a depth below 30 ft., the lower aquifer did not contain detectable contaminants. The same values were used with an average dissolved lead concentration of 1.8 ug/L, resulting in an average mass loading below 18 mg/day lead. However, the actual loadings to the lake will be less than this value since several interfering mechanisms take place between the well point and the water discharge to the lake. PCBs and lead exhibit particularly strong affinities for soil and suspended particulates, and would not be expected to migrate significantly either vertically or horizontally. A significant percentage of PCB residues, if these were actually

present in ground water, will be removed by soils; dissolved lead species may also precipitate by interaction with soil matter to reduce the levels reaching the lake. Furthermore, the contaminants will be diluted by several orders of magnitude upon reaching the lake, and some persistent residue levels may adsorb to suspended solids in the water column and eventually settle.

Based on the detection of minimal site contaminants in groundwater, limited transport as demonstrated above, and no ground water uses at this site, this exposure route is incomplete and will not be considered further in the risk assessment.

36.4.1.3 Receptor Evaluation

Human

The Area 9 Landfill is situated in a non-populated area adjacent to an industrial facility, and is fenced. Therefore, the only potential human receptors would include facility employees, site trespassers possibly including children, and (given the closeness of the site to Crab Orchard Lake) occasional recreational users of the Refuge. The number of human receptors is very low and exposure will be of a transient, non-chronic nature. Specific scenarios for human exposure to site indicator contaminants will be developed in the following sections. The transport route evaluation identified three functional transport mechanisms: the air route, the direct contact route, and the surface water transport route. Exposures will generally occur only in the vicinity of the landfill, with the possible exception of downgradient creeks and lowlands near the lake, and consumption of fish taken from the lake. The following are the most likely human exposure scenarios for the functional transport routes.

a) Direct Contact: The most probable human exposure scenario would be exposure to site indicators via direct contact with surface residues at the landfill and to sediments in the several intermittent creeks that drain the landfill. These exposures will be on a one time- or very limited basis and of a short term nature. The small number of Olin facility employees constitute one group of potential receptors. Trespassers gaining access to the fenced landfill area or hikers traversing contaminated creek sediments may also be exposed to contaminants via direct contact. Although the landfill is generally well vegetated, there are areas of exposed wastes and soils that could serve as a potential source of exposure. The most likely mode of entry of contaminants into the body would be incidental ingestion of soil-bound residues adhering to the skin, clothing, or shoes, acquired by direct contact with exposed wastes.

b) Air Route: Dusts generated by wind erosion or foot traffic over exposed waste areas constitute the major source and mechanism for exposure via the air route since the site contaminants are soil-bound and non-volatile. As with the direct contact pathway, the receptors include facility employees and other passersby who may breathe contaminated dusts while traversing the landfill. These exposures will also be acute in nature.

c) Surface Water Route: Monitoring of sediments of intermittent creeks draining the landfill indicate that low concentrations of site contaminants have migrated from the landfill, presumably by surface water runoff. The presence of PCB residues in sediments of Crab Orchard Lake near the Area 9 Landfill site (Section 2.6.1) suggests that transport of contaminants to the lake is occurring. Therefore,

some degree of exposure is possible from the consumption of fish taken from the vicinity of the source which have accumulated site residues in edible tissues.

d) As presented in the preceding section, transport of site contaminants to Crab Orchard Lake sediments presents a functional human exposure pathway via ingestion of residues accumulated in fish.

Wildlife

Due to the industrial nature of this site, the diversity and abundance of habitat may be relatively less than at other, less restricted areas of the Refuge. Nevertheless, the vegetated landfill provides adequate habitat for invertebrates and small vertebrates, and the nearness of Crab Orchard Lake creates the opportunity for exposures to aquatic populations. Occassionally, deer have been seen within the landfill area; however, these animals have not shown any impact from PCBs or lead at the site based on studies summarized in Section 2.7.

a) Direct Contact: Wildlife inhabiting the landfill such as invertebrates and small burrowing rodents will receive both acute and chronic direct contact exposures to site contaminants bound to soil dusts during burrowing activities. Exposed waste areas present a functional direct contact route exposure path for birds while feeding, ingesting grit, and dusting. Subsequent ingestion of soil-bound residues while preening or grooming is the principal means of entry into the body.

b) Air Route: Inhalation exposures of wildlife to dust-bound site contaminants will follow the direct contact scenario described above. Exposures to PCBs due to their volatilization from soils were evaluated in Section 24.4.2.1, and also contribute to this exposure

route, especially for small rodents which would be in intimate contact with soils during extended periods (e.g. within a burrow).

c) Surface Water: Transport of site contaminants to Crab Orchard Lake from the landfill via runoff of sediments creates a functional chronic exposure pathway for aquatic organisms in the bay area of Crab Orchard Lake. Exposures will be relatively greatest for benthic invertebrates and bottom-feeding fishes such as catfish.

d) Ingestion: Implied in all three wildlife exposure pathways discussed above is the ingestion of site contaminants via soils, dusts, sediments, and prey consumed in various activities. In addition, herbivores may consume contaminated dusts on seeds and vegetation and fish, birds (i.e. ducks, herons) and other aquatic organisms may inadvertently ingest contaminant-bearing sediments while feeding. Site contaminants, especially PCBs, are prone to accumulate in aquatic food chains, adding to the importance of the ingestion route of exposure.

36.4.2 Quantitative Assessment

Because of the relatively greater magnitude of PCB residues at this site compared to lead, and because of potential for non-threshold toxicity presented, the quantitative assessment will focus primarily on PCBs as the site contaminants of greatest concern. Exposure to lead residues will be addressed due to the presence of lead in non-fenced areas of the site.

36.4.2.1 Estimates of Release and Exposure Rates

Estimates of Exposures by Direct Contact

The qualitative assessment for the Area 9 Landfill has determined that direct contact represents a functional exposure pathway for humans and wildlife. However, PCBs and lead are tightly bound to soils and sediments, and are poorly absorbed through the skin. Therefore, dermal absorption of contaminants is not expected. The scenario consists, instead, of ingestion of bound residues picked up through direct contact with soils and sediments. Therefore, the contribution of this route of exposure will be addressed in the section below on ingestion exposures.

Estimates of Airborne Exposures

The qualitative portion of this assessment has established that the air pathway represents a complete exposure route. The pathway consists of breathing of contaminated dusts at the landfill site by occasional human activities (visits by employees of the adjacent Olin facility, trespassers, etc.), and by burrowing and dusting activities of wildlife. The general approach and assumptions used to estimate airborne human and wildlife exposures is given in Section 24.4.2.1 of this report. Using this approach for a four hour excursion by a facility employee or trespasser in a sector of the landfill containing exposed wastes, and assuming a mean surface PCB soil level of 3,200 mg/kg, a total exposure to PCB of 166 ug/day, or 2.4 ug/kg body weight is obtained as an inhalation exposure rate. Assuming three such visits to the site per year, a chronic inhalation rate of 0.02 ug/kg/day is derived. It should be realized that such a scenario does not technically define a chronic exposure, and is essentially invalid on toxicological grounds. However, regulatory agencies currently suggest this approach to estimate the "worst case". The acute worst-case inhalation exposure level of 2.4 ug/kg is far lower than would create any concern for acute toxicity. Using a representative intermediate lead soil

concentration of 4,000 mg/kg, a one day exposure of 3.0 ug/kg is estimated under the same worst case conditions. The contribution of inhaled residues to total acute intake is discussed in Section 36.4.2.2. True chronic inhalation exposures are likely, however, for small burrowing mammals at the site. Using a breathing rate value of 0.006 m3/hour for a 30 g mouse (U.S. EPA 1985) and creation of a 10 mg/m3 dust containing 3,200 mg/kg of PCBs during 1 hour of daily burrowing, a daily chronic exposure of 6.4 ug/kg/day is obtained for inhalation of dusts. In addition, although PCBs exhibit a low vapor pressure it will be assumed that at the concentrations found in surface soils the air is saturated with PCB vapors. This assumption would not likely hold at low levels of PCBs in soil due to the adsorptive forces of these compounds with soil. Small rodents living in burrow areas at the site would inhale these saturated vapors. The assumptions used to derive an exposure level for PCB vapors were detailed in Section 24.4.2.2. Applying a similar rationale for rodents living at the Area 9 Landfill, mice might inhale 1.09 mg/kg/day if exposed to saturated vapors during 16 hours inside a burrow. The total inhalation exposure from PCBs is thus (1.09 + 0.0064) = 1.1 mg/kg/day.

Exposure to lead, on the other hand, would be mainly through inhalation of dusts during burrowing. For lead concentrations in soil/sediment at 4,000 mg/kg, the exposure rate would be 5.6 ug/kg/day. The significance of these exposures is discussed later in Section 36.4.2.2.

Estimates of Ground Water Exposures

It has previously been determined that the groundwater exposure pathway is incomplete at the Area 9 Landfill and therefore will not be considered quantitatively.

Estimates of Surface Water Exposures

In view of a functional transport mechanism for conveying site contaminants to Crab Orchard Lake via runoff events, the surface water pathway is complete. The actual mechanism of exposure will be via ingestion of contaminants accumulated from residues present in sediments of a bay of the lake adjacent to the landfill. Therefore, exposures by this route will be discussed in the following section on ingestion exposure.

Estimates of Ingestion Exposures

Ingestion exposure of site contaminants at the Area 9 Landfill and contiguous sites has two components: ingestion of soil-bound residues acquired by direct contact with waste materials, and bioconcentration and foodchain accumulation of PCBs in a nearby bay of Crab Orchard Lake, with potential exposures to humans and terrestrial wildlife consuming contaminated aquatic organisms. The basic approach and assumptions used to estimate human and wildlife exposures by direct contact and ingestion of contaminated soils have been discussed in Section 24.4.2.1, the quantitative assessment for the Job Corps site. Using the worst case assumption that an individual ingests 100 mg of soil as a result of an excursion into an exposed waste area of the Area 9 Landfill, and that the mean surface level of PCBs is 3,200 mg/kg, an ingestion rate of 4.6 ug/kg/visit is estimated for a 70 kg adult. Using the same assumptions and a exposed surface soil lead concentration of 4,000 mg/kg, an acute ingestion of lead is estimated at 5.7 ug/kg/visit. Wildlife PCB intake rates from ingestion of contaminated soil at the site during feeding or grooming are detailed in Section 36.4.2.2, Quantitative Assessment.

Estimates of intakes to piscivorous mammals such as mink and otter, which could be exposed to residues of PCBs from ingestion of fish in the area of the Area 9 Embayment, are presented in Section 38.4, Environmental Effects for Crab Orchard Lake. Exposures to fish-eating birds (bald-eagle, osprey, duck) is also addressed in Section 38.4.

Ingestion of residues found in the water of the Area 9 Embayment are possible for terrestrial mammals. Water intake rates for such species will be assumed to be 10 percent of body weight for hervibores and 30 percent of body weight for carnivores. The PCB concentration in a water column sample taken from the lake, just outside of the embayment was 19 ng/L (ppt). Assuming this concentration is representative of the bay area water, it is estimated that the three hervibores evaluated (mallard duck, rabbit and mouse) each receive an exposure of 1.9 ng/kg/day.

36.4.2.2 Quantitative Risk Assessment

Human Risks

Human exposure at the Area 9 Landfill will be extremely limited. With the installation of a chain link fence around the landfill in 1984, the direct contact and airborne pathways described above may very well be incomplete. Visits by Olin employees would be presumably authorized with appropriate protective equipment used, and excursions by hunters and hikers are eliminated. Only intentional trespassing onto the landfill would provide a complete pathway. Human exposure estimates for airborne dust-bound PCB residues (2.4 ug/kg/day) and direct contact ingestion of soil-bound residues of PCBs (4.6 ug/kg/visit) provide a total estimate of 7.0 ug/kg/visit for this scenario. This intake is far lower than any that would be cause for concern from acute PCB toxicity. Chronic PCB

exposures are not possible for this site and it would be inappropriate to derive estimates for such.

U.S. EPA (1987) developed a drinking water health advisory, however, for short term exposure to PCBs (Aroclor 1254) in drinking water. Selecting a representative no observed adverse effect level of 1 mg/kg/day based on effects on liver weight in rats exposed for seven days, an acceptable short term, 10 day, exposure level in humans of 0.7 mg/day, or 10 ug/kg/day for a 70 kg adult, was derived. Apportioning the upper level acute exposure estimate for a trespasser within the Area 9 landfill over a period of 10 days, results in a daily intake of 0.26 ug/kg/day, a level within the short term acceptable exposure limit. This estimated intake level assumes that a trespasser may be exposed as a result of one excursion during this 10 day period, since, due to the presence of a tall locked chain-link fence surrounding the property, successive daily visits are not reasonably assumed at this site.

Similarly, for lead, exposure to surface soils containing 4000 mg/kg by the ingestion and inhalation routes would produce a total acute intake of 8.7 ug/kg/visit. In comparison, U.S. EPA (1980) established an acceptable daily intake of 1.4 ug/kg/day for chronic lead ingestion in drinking water. Therefore, acute lead exposures under worst case site conditions would pose minimal risk.

An additional avenue for human exposures is consumption of fish taken from the bay which connects with Crab Orchard Lake. PCB-contaminated sediments provide a source for bioaccumulation of residues. Associated risk levels due to consumption of fish from Crab Orchard Lake are presented in Section 38.4, Environmental Effects for Crab Orchard Lake.

Restricting access to the landfill also reduces the potential for deer to acquire site contaminant residues which could be ingested with venison. Although deer have been seen at the landfill, previous studies (see Section 2.7) have shown no contamination in deer hunted in the vicinity of this site.

Wildlife Risks

Estimates of total PCB intakes in receptor species of wildlife are given below:

Estimated Daily PCB Intake

Species	Body Weight kg	Inhalation (mg/kg/day)	Ingestion Food (mg/kg/day)	Ingestion Water (ng/kg/day)	Total Exposure mg/kg/day
Mallard	1.0	NA	30.4	1.9	30.4
Rabbit	1.0	0.026	45.6	1.9	45.6
Mouse	0.03	1.09	30.4	1.9	31.5

Note: See Table 24-1, Section 24.4.2.2 for assumptions.

NA = not applicable

These estimates indicate that wildlife exposures to site-related PCB residues may be greatest among small mammals on the landfill such as rabbit, mice, chipmunks, and the like. These animals will be exposed primarily via ingestion of contaminated soil and vegetation dust while burrowing, grooming, and feeding on dust-bearing seeds and invertebrates. Given the broad range of demonstrated possible PCB effects, the potential for interspecific sensitivity, and limited data on effects of PCBs on wildlife species, it is difficult to gauge the significance of these exposures. Using data from controlled tests with laboratory strains of animals, intake levels as high as 21000 ug/kg/day may not be sufficient to induce liver pathology, reproductive success, teratogenic effects, or neoplasia (U.S. EPA 1980). However, the

possibility for behavioural, immunological, or other subtle effects at these exposure levels which decrease competitiveness and survival in wild species cannot be discounted. Also, assuming that ingested PCBs are accumulated in these small mammals to some degree, a complete, non-quantifiable exposure pathway may exist for small predators such as skunks, weasels, and hawks which are able to gain access to the landfill while hunting.

Lead exposure in small burrowing mammals and other terrestrial wildlife may also present risks from chronic effects. For instance, using the previous exposure scenarios and a 4,000 mg/kg lead soil content, a chronic intake of 38 mg/kg/day is estimated for a mouse. These exposure levels could very likely produce behavioural, reproductive and other chronic effects if maintained at the assumed levels.

In contrast, exposure of aquatic wildlife or organisms feeding in the lake bay adjoining the landfill appear to be low. Ingestion rates of PCBs estimated to occur by consumption of contaminated fish by piscivorous birds and mammals are discussed in Section 38.4. Adverse effects to benthic organisms or species using PCB-contaminated sediments as a spawning substrate are possible but cannot be predicted from available data.

36.4.3 Analysis of Uncertainties

Among the areas of uncertainty in this analysis are the adequacy of the analytical data base and the set of assumptions that were selected to analyze human risk at the site. PCB analyses were sufficient to derive a representative soil level for the exposure assessment. However, lead content of site soil was extremely variable, with most sites showing only

background values. In order to conservatively estimate potential risks, a soil lead content of 4000 mg/kg was selected as representative, based on a range of samples with elevated lead of 205 to 8270 mg/kg. Other assumptions were also biased towards an upper bound worst case, in order to be most protective of public health. Some of these include consideration of trespassing across a chain link fence around the site, the presence of significant amounts of exposed contaminated soil, and ingestion of very large amounts of soil during these incursions. TCDF was not chosen as a site indicator contaminant for analysis due to lack of information on whether a toxic isomer of TCDF, such as the 2,3,7,8- isomer, was present.

Further, there is no evidence at present that TCDF poses a non-threshold carcinogenic risk comparable to PCBs, or that significant amounts of TCDFs bound to soil can be absorbed if ingested (Paustenbach, 1986). If this situation should change in the future, a revised risk assessment should be made.

Two other principal areas of uncertainty exist in addressing the risks posed to wildlife by chronic exposure to PCBs at the landfill site. A lack of documentation on the effects of PCBs on wildlife species which might be found on the site necessitated the use of studies involving laboratory rodents and rabbits as surrogates. The relative sensitivity of these species is unknown. Similarly, the chronic effects of benthic aquatic organisms to residue levels of PCBs found in the bay connecting with Crab Orchard Lake are largely unknown. Virtually all controlled toxicity assays are performed with toxicant in the water column only, which may be inadequate to determine the risks to benthic organisms exposed to contaminants in bottom sediments and interstitial waters.

Two main problems with conducting assays with non-water soluble contaminants are: 1)the water does not acquire a high enough concentration of the contaminant to indicate that there is a problem, usually measured by mortality rates; and 2)bioassays are short term, 96 hours or less, and severe chronic impacts cannot be measured in this short time span. Interstitial water spun off in a centrifuge is now being used in bioassays to determine the toxicity of non-water soluble contaminants.

36.5 Preliminary Remedial Alternatives

The sampling data and the risk assessment described in the previous sections indicate the presence of contaminants in soil which will require remediation. Contaminated areas containing PCB levels above 50 mg/kg are have been highlighted in Figure 37-4 and include some soil pockets up to 12 feet depth in the landfill. The sediments in the lowland area northeast of the landfill contained PCB levels below 5 mg/kg, with the exception of three samples (see Figure 36-3). The samples collected from the lake embayment adjacent to the site contained PCBs below 1 mg/kg with the exception of one sample (4.09 mg/Kg) taken close to the shoreline. Remediation of the site might address alternative measures for controlling potential leaching of contaminants to surface runoff and lake waters.

The criterion for cleanup for PCB contamination in surface or drinking waters, grazing lands, and vegetable gardens is set by EPA on a site specific basis and will be addressed in the FS. Based on the assumptions developed in Section 36.4, the contaminant levels do not pose long-term risks to chronically exposed wildlife or repetitive site visitors. It should be noted that these long-term, repetitive exposures are highly unlikely for humans, since the site

is somewhat isolated and site access is restricted by a tall fence and locked gates. However, if this scenario were possible, it is estimated that the risk levels to humans due to PCBs could be 40 to 400 fold greater than would be considered acceptable. Thus, reducing exposure to surface PCB levels in soil to between 7 to 70 mg/kg will reduce risks to an acceptable range of 10^{-6} to 10^{-5} for humans. This level is also 14 to 1.4 lower in magnitude than the no observed effect level for protection of the most sensitive species of wildlife from chronic effects (see Section 24.4.2.2). Exposures to wildlife under an example cleanup level of 50 mg/kg PCBs were presented in Section 24.5.

Other contaminants detected in the soils above the levels for Refuge background include lead and mercury; however, remediation of the site for PCBs will also include those areas contaminated with lead or mercury. The ground water samples contained traces of PCBs below 0.045 ug/L. Total chromium in one well sample exceeded the standard but the corresponding dissolved concentration was below the standard. Some alternatives for remediation of contaminated Lake water in the embayment area adjacent to the Landfill might also be addressed as part of the remedial efforts for this site. Lake samples offshore from the Area 9 embayment (see Section 38) contained PCBs in the range of 0.008 to 0.019 ug/L in water columns and up to 77 ug/kg in sediment samples; however, only one water sample exceeded the ambient water criteria for aquatic life protection of 0.014 ug/L (24-hour average). All water samples were below the AWQC acute toxicity level of 2.0 ug/L for protection of aquatic life.

A summary of the potential remedial response actions and associated technologies for this site is presented in Table 2 of the Executive Summary. Remedial measures including excavation, capping, regrading, revegetating, and

surface water diversion will be evaluated in the FS for this site. A brief discussion of the potential responses is included below.

Limited Site Access

Currently the site is enclosed by a tall fence which remains locked at all times. Access to the site is further limited by a locked gate at the entrance of the access road to the Landfill. Continued site access limitations will greatly reduce the potential for human and/or wildlife exposure via the water or direct contact with soils/sediments. The site may require site use limitation until all contaminated materials have been removed or adequately contained.

Surface Water Control

Dikes or trenches for diversion of surface runoff could be constructed in the lowland area and around the landfill to preclude contamination of lake waters.

Removal or Containment of Soil and Sediment

Contaminated soil might be excavated and removed for treatment on-site or off-site or regraded and contained on-site in a secure landfill.

Based on the sampling results from this RI, dependent on depth of contamination, soil to a depth of 6 or 12 ft may require removal or containment. Containment or removal of selected sediments will reduce the potential for surface and ground water contamination. Clean fill will be used for regrading and capping. Several abandoned munitions storage bunkers on the Refuge could be used as industrial cells if retrofitted for containment of treated wastes.

On-Site or Off-Site Treatment

Technologies such as incineration, vitrification, solidification or fixation should be considered for PCB contaminated soils.

Monitoring

The remedial response might include periodic sampling and analyses of soil/sediments and of five site monitoring wells for chromium (waters only), lead and PCBs. Follow-up studies might begin immediately after remediation to verify the adequacy of the cleanup.

36.6 Conclusions and Recommendations

It can be concluded that the Area 9 Landfill Site is impacted, with the primary pollutants being PCBs, lead, and mercury. Exposure to the site has been minimized by a chain-link fence. It is recommended that remedial alternatives for this site be evaluated in the FS. Potentially applicable remedial measures include removal or containment of contaminated soil and sediments, as well as ground water and surface water monitoring.

SECTION 37 - SITE 33, AREA 9 BUILDING COMPLEX

37.1 Site Description

Further information on Area 9 can be found in Section 36.1. Site 33 consists of the Area 9 Building Complex currently occupied by Olin Corporation and used primarily for the manufacture of explosives. (See Figure 37.1). From 1946 to 1962, Site 33 was occupied by Sangamo Weston, Inc., Capacitor Division to manufacture power factor capacitors, AC motor run capacitors, and a variety of DC capacitors.

The components utilized by Sangamo in its operations were of various types and included aluminum, electrolytes, mica, and silver and lead foil. The Division also manufactured small transformers that used mineral oil as a dielectric. Subsequently, Olin Corporation has been using the industrial facilities at the site to manufacture explosives.

Previous soil investigations at Area 9 include one study conducted by Envirodyne Engineers for the Olin Corporation in 1984. The focus of that study was to identify the extent of PCB contamination in soil. The analytical results (PCB analyses) are presented in Figure 37-1. The soil samples collected adjacent to Buildings I-1-23 and I-1-2 contained PCB levels above 50 mg/kg wet weight. Some isolated samples collected along the sides of the access road to the Area 9 Landfill also contained elevated PCB concentrations.

37.2 Site Investigations

37.2.1 Phase I Site Investigations:

The objective of the Phase I soil sampling was to define the horizontal and vertical limits of contamination. The sampling locations were selected based on data from previous investigations for Olin Corporation and

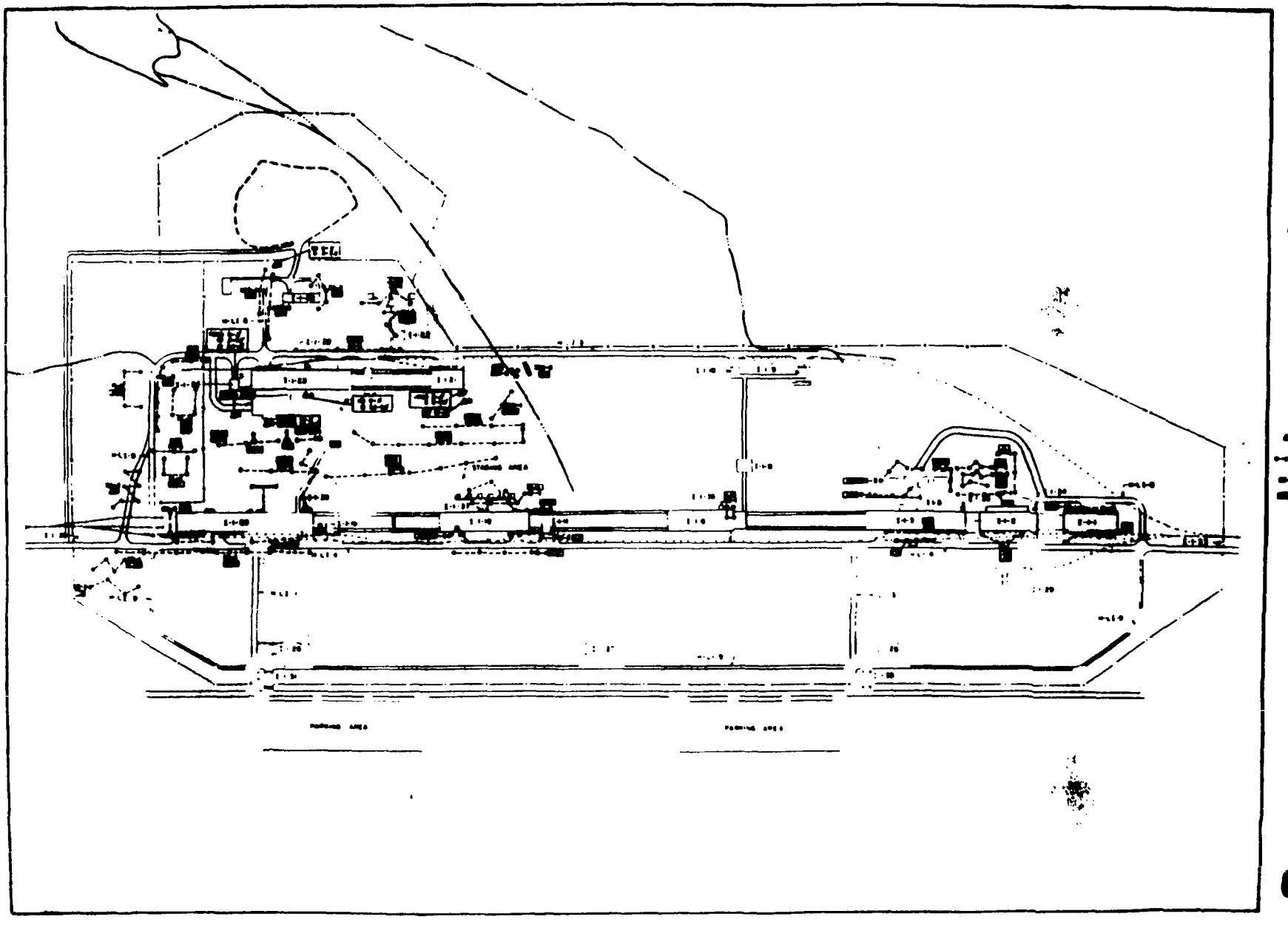


FIGURE 27-1

SITE 33

AREA 9 BUILDING COMPLEX
(OLD DATA)

included, drainage pathways (located from aerial photographs), locations in close proximity to buildings, and transportation routes used to dispose of solid wastes. A total of 188 individual soil samples were collected, including surface, 0-1 ft., 1-2 ft., and 2-3 ft. depths. These samples represented 102 distinct locations within the Complex. (See Figure 37-2).

37.2.2 Phase II Site Investigations:

Additional soil samples were collected from 61 new locations and at greater depths (up to 6 ft.) for selected Phase I locations. Phase II was performed to more clearly define the extent of contamination. These samples were collected adjacent to contaminated areas identified in Phase I, at deeper locations, or from areas downgradient of Phase I samples, and also from several drainage ditches to trace the extent of migration. (See Figure 37-3). The soil samples were analyzed for PCBs; in addition, three of these soils were analyzed for the full list of priority pollutants. Three ground water monitoring wells were installed and sampled. These wells were installed to depths between 14 and 20 feet within silt and fine sand soils and were screened at the lower five feet of the boring. The ground water samples were analyzed for the full list of CLP organics, nitrosoamines, explosives, metals, and cyanide.

37.2.3 Site Hydrogeologic Characterization

37.2.3.1 Site Geology

Information obtained from the subsurface soil boring and wel installation program indicates that the site is immediately underlain by silty clay at least 25 feet thick. Since deeper well installations were not performed at this site, it is not possible to describe with any degree of

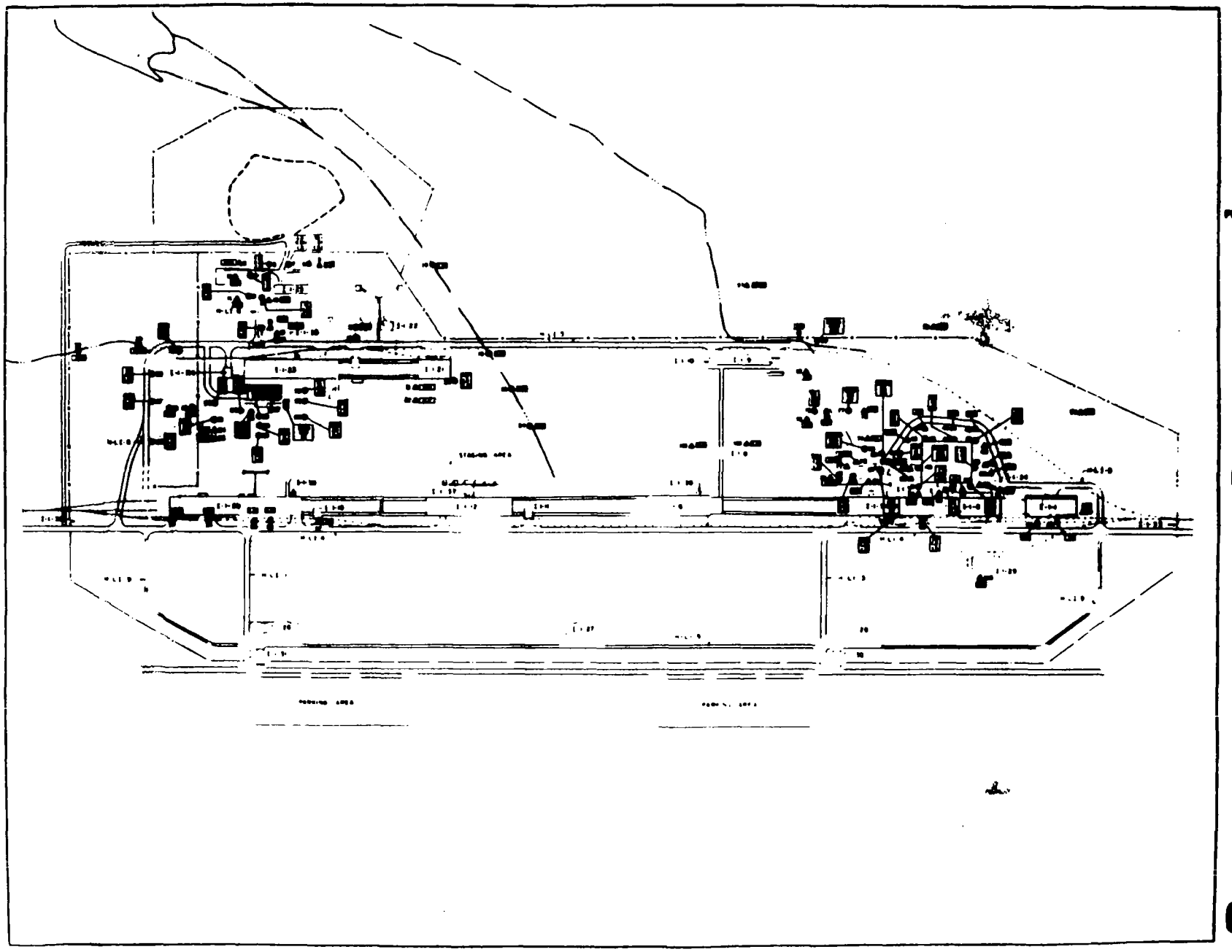


FIGURE 37-2

SITE 33

PHASE I SAMPLING LOCATI

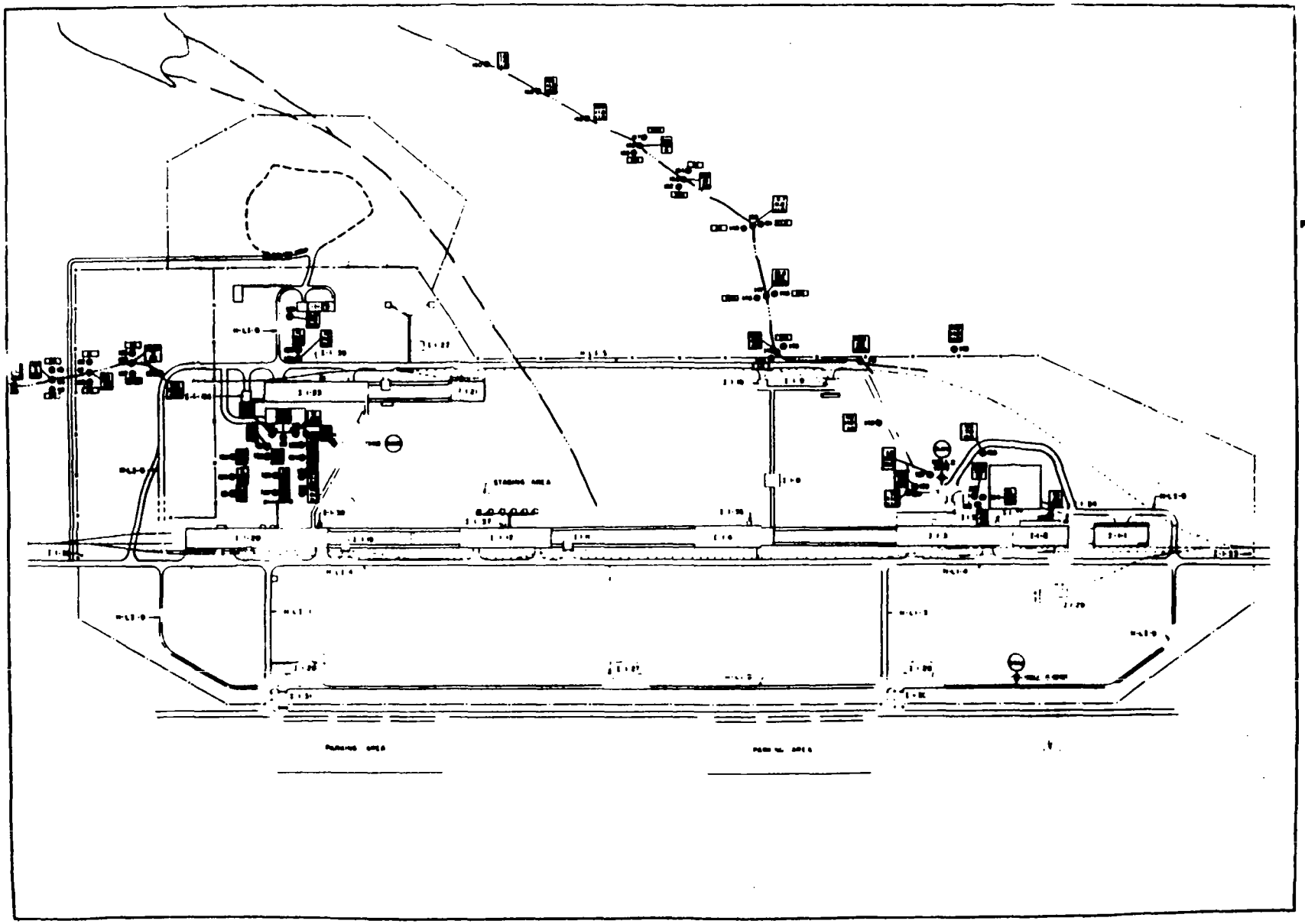


FIGURE 37-6

SITE 33

PHASE II SAMPLING LOCATIONS

certainty the soil types and thicknesses occurring beneath the silty clay. However, information from nearby Site 36 (Area 9 Landfill) indicates that the sand layers occurring there may continue beneath this site. Similarly, although bedrock was not encountered during drilling at this site, extrapolation of that surface between known data points on hydrogeologic cross-section A - A' (Figure 4-2) indicates bedrock may occur at a depth of 70 feet.

37.2.3.2 Site Hydrogeology

Occurrence of Ground Water

Ground water occurring within the three site wells which screen the shallow ground water table indicate an upper water table occurs about 6 to 14 feet below ground surface. Ground water elevation data collected in June 1986 indicates a higher water table of 5 to 8 feet occured during that period. Since no deeper wells were installed at this site, the nature of the ground water occurring in deeper portions of the unconsolidated sequence could not be evaluated.

Ground Water Flow Conditions

Ground water elevations obtained from the site wells on June 18-19, 1987 were contoured and are shown on Figure 36-4. Ground water flow directions interpreted from the figure indicate flow to occur both northeast and southwest off of a divide located near the southeastern end of the Olin Complex. The reason for the reversal of ground water flow to the southwest is most likely the presence of a shallow intermittent stream and swale located in this area. Surface water receives ground

water discharge from the southwestern portion of the site, then flows north into Crab Orchard Lake.

Ground water flow velocity in the northeast direction toward Crab Orchard Lake was calculated for this site. No velocity calculations could be made to the southwest due to limited data points. The hydraulic gradient (I) to the northeast averaged about 0.011 ft/ft. The average hydraulic conductivity (K) from wells 33-341 and 33-342 was calculated to be 1.3×10^{-5} ft/sec, or 1.12 ft/day. Porosity was assumed to be about 0.035 ft/day, or 12.8 ft/year, from data collected in June 1987.

37.3 Analytical Results (See Appendix I, Page 30)

37.3.1 Phase I Analytical Results:

Figure 37-2 shows the sampling locations and detected PCB levels in soils. The results supported the findings of Olin Corporation, in which the most significant locations for PCB contamination are adjacent to Buildings I-1-2 and I-1-23 where PCB concentrations exceeded 1,500 mg/kg. The results also identified two drainage ditches where contaminated storm runoff or spills have caused PCB migration from the site. Apparently, these drainage routes receive runoff water from areas surrounding Buildings I-1-2 and I-1-23.

A third drainage ditch originating in the Complex which passes east of the landfill and discharges to Crab Orchard Lake did not show evidence of contamination (PCB concentrations below 1 mg/kg). The old roadway access from the Building Complex to the landfill showed only isolated sampling locations where PCB concentrations exceeded 25 mg/kg. The locations where PCB levels exceeded 25 mg/kg were limited to the surface and/or upper 1 ft. of soil near the landfill access.

The soils collected along the west face of Building I-1-23 ranged from 500 to 120,000 mg/kg wet weight PCBs at the surface. Three soil cores collected within 30 ft. of the building contained elevated PCB levels to a depth of 3 ft. PCB concentrations in most samples collected from the lawn further from the building exceeded 50 mg/kg at the surface (0-1 ft.), but were below that criterion in the subsurface (1-3 ft. depth) samples. Two surface (0-1 ft depth) samples collected away from the building along the drainage ditch to the north contained PCB concentrations of 1200 and 1300 mg/kg wet weight.

The extent of PCB contamination (above 50 mg/kg wet weight) in the vicinity of Building I-1-2 was, for the most part, limited to the immediate areas surrounding the building and up to the access road. Some grab samples collected from either side of the access road and one collected close to a drainage route toward the lake also contained PCB concentrations that exceeded 50 mg/kg. Four soil cores collected at a depth of 3 ft. contained PCB concentrations in excess of 50 mg/kg also.

The results for PCDD and PCDF isomers in soils are presented on page 30A of Appendix I. A separate data listing is included to illustrate the comparison of actual dioxin/furan to PCB ratios compared to the ratios which would be expected based on the corresponding PCB concentration for that sample. The purpose of calculating this ratio is to determine if dioxins/dibenzofurans are elevated due to burning products of PCBs. If this were the case, then the dioxin/furan to PCB ratios would be higher than the expected normal ratios for PCBs. The Table lists a concentration for each peak detected in the scan (isomer No.) and a total concentration for each compound. This total concentration was compared to the average PCB concentration detected in the same sample to develop a ratio of PCDD

or PCDF to PCBs. An average PCB concentration was used because the analyses were conducted in duplicate (one reported by ETC Laboratories and one reported by OBG Laboratories).

Based on studies conducted by T. Sawyer and S. Safe et.al.(1982, 1984, 1985) relative to PCB isomers and congeners, the expected fraction of PCDD and PCDF isomers associated with a measured PCB concentration can be calculated. The 'equivalent' fraction of dioxins and furans is determined by a conversion factor or Keq value. The Keq value is derived based on 1.00E-05 for 2,3,7,8-TCDD.

The results for Area 9 Building Complex showed positive detections above the ratios which would be expected for the sample collected in front of Building 1-1-23. The dibenzofuran isomers ranged from 28 to 249 ug/kg; the highest concentrations were associated with HxCDF (249 ug/kg), and PCDF (158 ug/kg), whereas the more toxic TCDF was detected at 28 ug/kg. The dioxins ranged from less than 0.11 for TCDD to 169 ug/kg for OCDD. The other soils contained traces of OCDD (6.2 to 9.7 ug/kg), and PCDF (less than 0.08 to 0.17 ug/kg) above the detection levels.

Four soil samples (0-1 ft. depth) screened for full priority pollutants did not contain any other organics at concentrations above the detection limits. However, the volatile and semi-volatile organic data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (See Exhibit B); some compounds which were not detected may in fact be present.

All measured concentrations for metals were similar to those detected at the control sites. The metals screening data are included in the data listing as estimated values.

37.9.2 Phase II Analytical Results:

Three ground water samples were collected at the Building Complex. PCBs were detected at concentrations of 0.093 ug/L (Well 33-340, adjacent to Building 1-1-23), 0.114 ug/L (Well 33-341, downgradient from Building 1-1-2), and 0.006 ug/L (Well 33-342, on the opposite end of the Complex, close to Building 1-1-28). These concentrations exceed the Ambient Water Quality Criteria for human health. In addition, wells 33-340 and 33-341 contained concentrations of chromium (113 ug/L and 50.0 ug/L respectively that exceeded the Illinois Public Water Supply Standards and Federal MCLs. However, the dissolved chromium levels below 1.3 and 1 ug/L for wells 33-340 and 33-341 respectively, were below the standards.

Traces of volatile organics were also detected in Well 33-341, including t-1,2-dichloroethene (404 ug/L) and trichloroethene (906 ug/L). The detected concentration of trichloroethene exceeded the AWQC of 18.4 ug/L for protection of human health. All other ground water parameters were below regulatory standards.

Three soils from the Phase II sampling were analyzed for full CLP organics and metals. Some organics (other than PCBs) were detected, including 1,2,4-trichlorobenzene (23,500 ug/kg in Sample 33-222), and 2-chloronaphthalene (6820 mg/kg in sample 33-270). Metal concentrations were generally within the ranges found at the control sites with the exception of one soil sample which contained 1400 ug/kg mercury (sample 33-291 at 8-1 ft. depth). Mercury was not elevated in the other soils analyzed for metals.

The detected PCB concentrations supported the findings from Phase I and provided the necessary data to better define the areas of contamination. (See wet weight PCB concentrations in Figure 37-3). The

results showed contamination along the two drainage paths from Buildings 1-1-23 and 1-1-2. Some samples contained levels above 50 mg/kg wet weight up to 3 feet depth, but PCB concentrations were well below that level in samples collected at further distances from the buildings. Core and surface sediment samples along the northeast drainage route downstream from Building 1-1-2 were taken up to where the creek enters Crab Orchard Lake. The last soil core (location 160) did not contain detectable PCBs to 1 mg/kg, from a high 4,100 mg/kg wet weight at the surface of the first soil core from the drainage path.

Soil cores to 3-ft. depths and surface samples were also taken from the drainage route from Building 1-1-23. The PCB levels along this drainage ditch ranged from 4,780 mg/kg to below 1 mg/kg wet weight. In addition, the surface sediments collected from the Area 9 embayment as part of Site 32 provided characterization for the area where the creeks entered the lake. Most sediments from the embayment contained PCB levels below 5 mg/kg wet weight, with the exception of two samples close to the lake boundary which contained 6.1 and 18 mg/kg PCBs. Figure 37-3 shows the sampling locations and the PCB concentrations along the ditches for the Building Complex. Soil cores collected from the lawn in front of Building 1-1-23 supported the Phase I findings and showed that PCB levels were elevated in the immediate vicinity of the building up to a depth of 6 ft.

The area around Building 1-1-23 is currently enclosed with a tall chain link fence. In addition, the immediate vicinity of the Building is roped off. The contaminated area is not mowed therefore the grass is tall and thick. PCB concentrations were well below 50 mg/kg in the soil samples (up to 3 foot depth) collected from the lawn area about 100 ft

from the building. Similarly, soil cores from the 3 ft. and 6 ft. depths adjacent to the concrete pad outside of Building 1-1-2 contained PCB levels that exceeded 50 mg/kg wet weight. The cores collected further from the building exhibited high PCB levels only at the 0-1 ft. depths. See wet weight PCB concentrations in Figures 37-3 and 37-4.

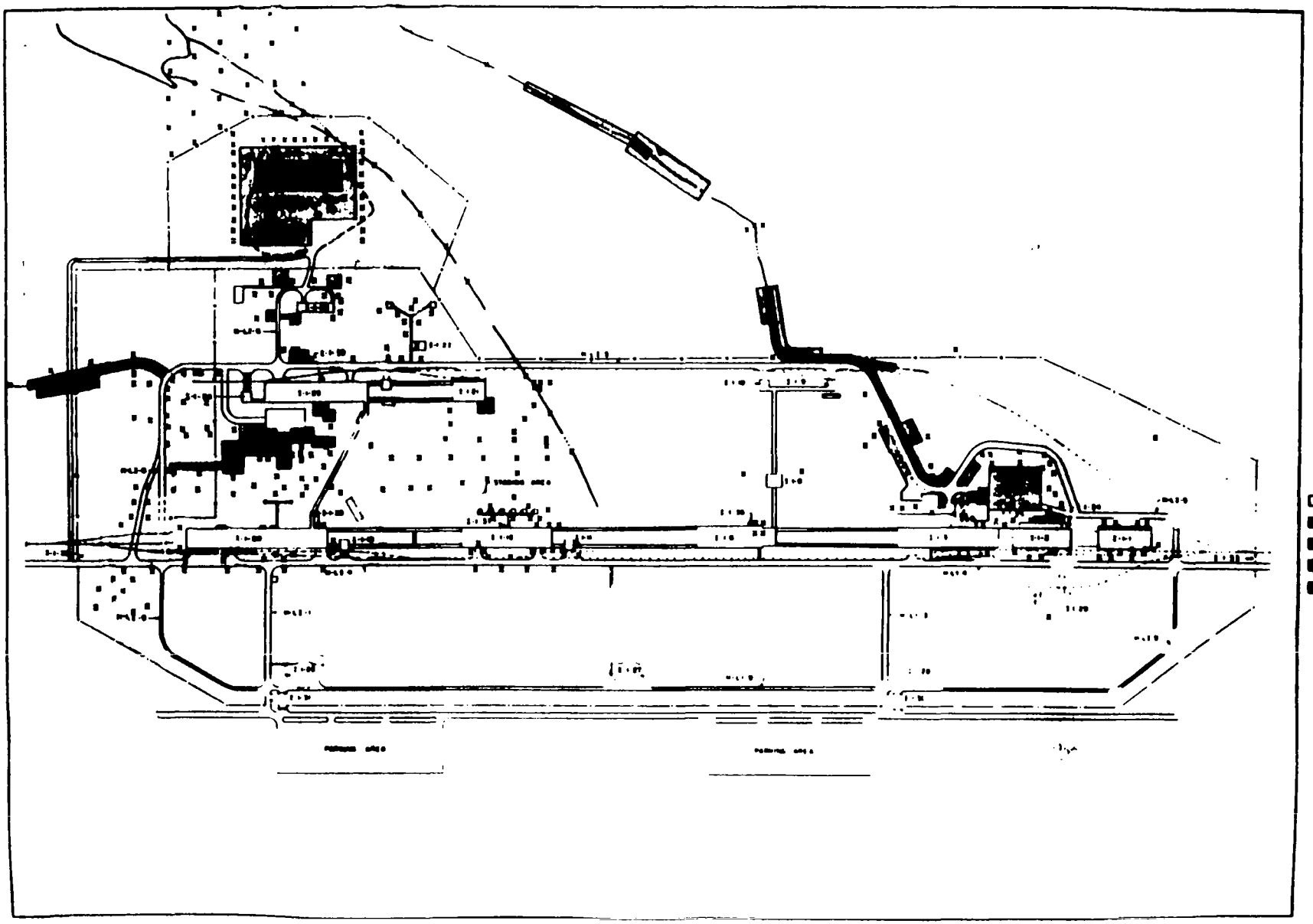
37.4 Environmental Effects

37.4.1 Qualitative Assessment

37.4.1.1 Source Evaluation

The results of the site investigations, as described in the preceding sections, determined that the Area 9 Building Complex has been used in the manufacture of capacitors, and explosives. Soil analyses of the site have shown the presence of several materials including PCBs, trichlorobenzene and chloronaphthalene. Three soil samples at the site contained tetrachlorodibenzofuran (TCDF), isomers unspecified, ranging from 0.14 ug/kg to 26.3 ug/kg. An unspecified isomer of tetrachlorodibenzodioxin (TCDD) was detected in one of these samples at 0.09 ug/kg. Trichloroethene, PCBs and t-1,2-dichloroethene were detected in site ground water at levels up to 906 ug/L, 0.114 ug/L, and 404 ug/L, respectively. No other contaminants in excess of Illinois General Use Water Supply Standards were found or any other water criteria or standards.

The areas contaminated with PCBs at concentrations above 50 mg/kg are identified in Figure 37-4. The total area has been estimated to be approximately 13 acres. The principal areas involved are immediately adjacent to Buildings 1-1-23 and 1-1-2, and two drainage paths leading from them (Figure 37-4).



SECTION 38 - SITE 34, CRAB ORCHARD LAKE

38.1 Site Description

Site 34 is comprised of Crab Orchard Lake (See Figure 38-1), which was formed in 1940 by construction of a spillway across Crab Orchard Creek. The lake has a surface area of 6,965 acres, a maximum depth of 30 feet, and 72,525 acre-feet of storage capacity. The retention time is 0.8 years (Kelly and Hite, 1981).

Water enters the lake through several creeks, including the Crab Orchard Creek at the eastern end of the lake. Water exits the lake through the spillway at the western end and through use of 280,000 gpd by the Refuge.

The eastern section of the lake has been bordered by manufacturing operations since the 1940s. Additional background on the site can be found in the introductory sections of this report. Previous investigations for water and sediments of Crab Orchard Lake are described in Section 2.6.

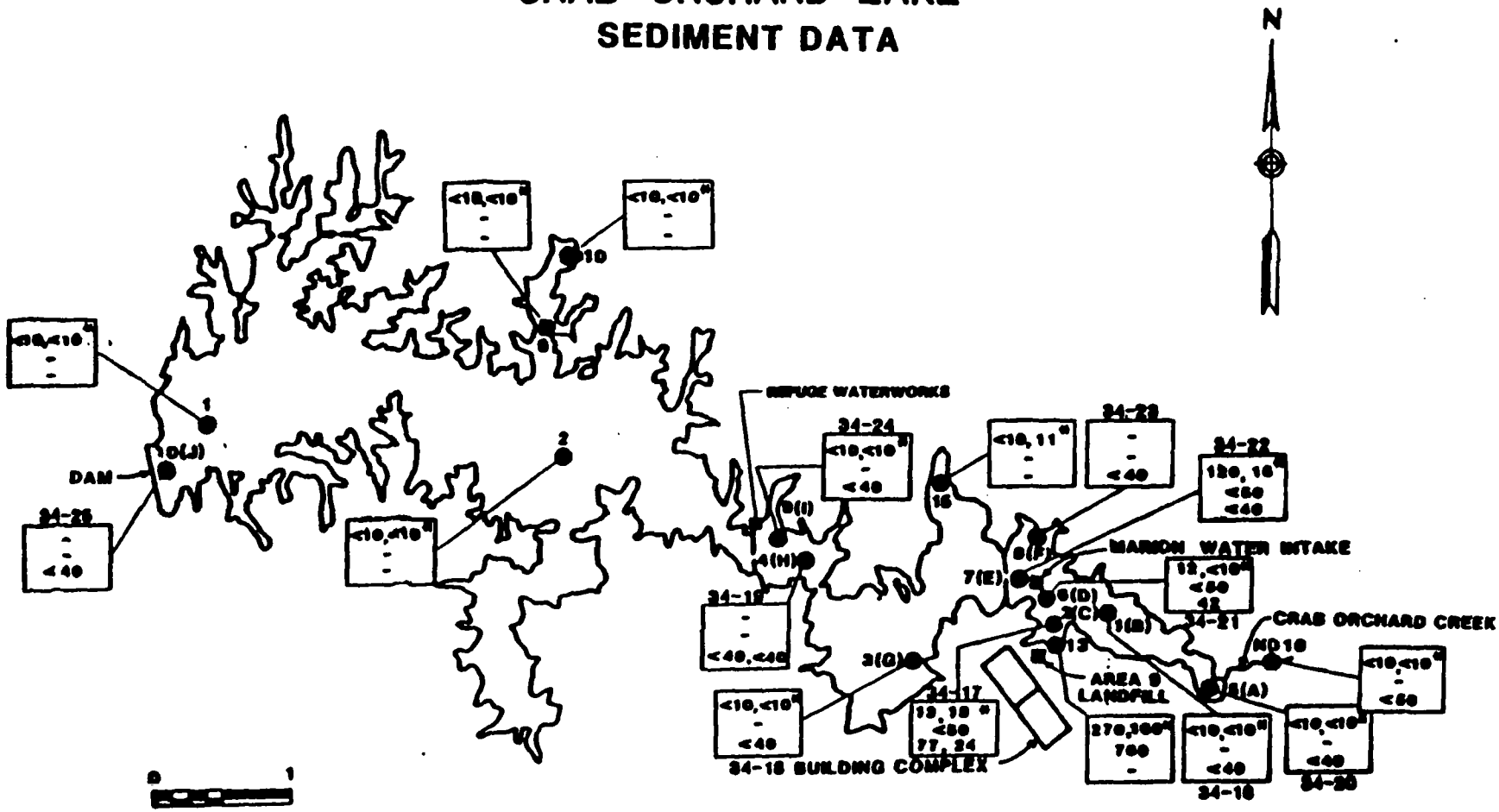
38.2 Site Investigations

38.2.1 Phase I Investigations:

Five water samples from current or potential drinking water sources were collected:

1. The Crab Orchard Refuge Treatment Plant Intake sample was collected at the intake structure under the grating and labeled as "Refuge Intake". The nearby Federal Penitentiary also derives its water supply from the Refuge water treatment plant.
2. The City of Marion WTP Intake sample at Marion Reservoir was taken by the dam at the intake structure and labeled as "Marion Intake".

CRAB ORCHARD LAKE
SEDIMENT DATA



LEGEND

● SAMPLE SITE

PCB CONCENTRATION, ug/kg (WET WEIGHT)

( )
( )
( )

MAY 25-26,1983 SITE, 1984)
MAY 21,1984 SITE, 1984)
DECEMBER 11-16,1986 – PHASE II SAMPLING

* REPLICATE SAMPLES

O'BRIEN & GERE ENGINEERS, INC

3. The Marion Reservoir Auxiliary Intake sample at Crab Orchard Lake was taken from the lake prior to the intake structure; it is referred to as the "Marion Reservoir Intake."

4. The Refuge Finished Water sample was taken from the tap at the Refuge Fire Station and labeled "Refuge Treated" (also treated supply for Penitentiary).

5. The Marion Finished Water was sampled from the city water tap, and was labeled "Marion Treated".

These samples were analyzed for drinking water quality parameters and PCBs. The Marion Reservoir Auxiliary Intake has historically been used only once or twice within the last fifteen years, as stated in Section 2.5. According to the City Engineer, the City of Marion currently uses Herrin Lake as a backup supply, rather than Crab Orchard Lake. Crab Orchard Lake will no longer be used to supplement the Marion Reservoir except as a last alternative. It is thus not a current drinking water source, but a potential drinking water source.

Thirty fish composite samples were collected from Crab Orchard Lake including bass, carp, bullhead, and catfish species. Fish samples consisted of single species composites of the edible tissue portions of two to five fish. The fish sampling locations are shown on Figure 38-3. The analyses of the fish samples were performed as part of the Phase II investigation.

38.2.2 Phase II Site Investigations:

The five current or potential public water sources were resampled and analyzed for nitrosamines, PCBs, metals and cyanide. Ten composite water columns were collected from the lake at three depths (see locations in

Figure 38-2): at the surface, mid-depth, and near the bottom. These samples were analyzed for cadmium, chromium, lead, arsenic, cyanide, PCBs, and low-level nitrosamines. Ten grab sediment samples were collected and analyzed for semi-volatiles, pesticides, PCBs, metals, and cyanide. Thirty (30) fish composite samples collected in Phase I were analyzed during the Phase II investigations for pesticides, PCBs, lead, mercury and cadmium. Each composite consisted of two to five individual fish of a particular species, including composites of bass, bullhead, carp, and catfish. The analytical procedures were performed on a homogenized composite of the edible tissues for each sample point. Specific procedures for filleting (skin-on, skin-off), blending, and digesting were detailed in Addendum No. 3 to the QAPP, March, 1987.

38.3 Analytical Results (See Appendix I, Page 32)

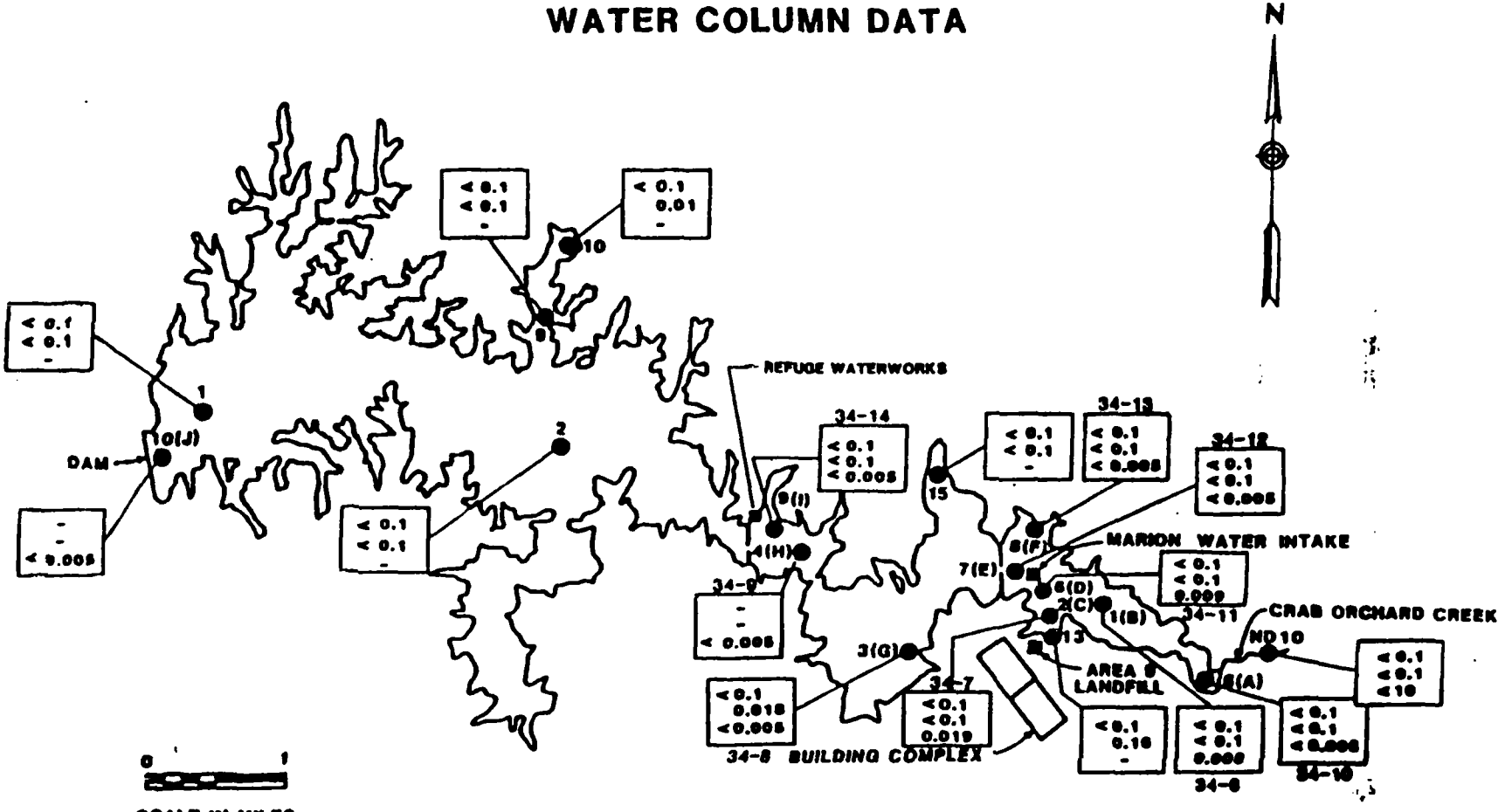
38.3.1 Phase I Analytical Results:

All the concentrations of detected contaminants in the current or potential water supply samples were below the Illinois Public Water Supply Standards and Federal drinking water standards with the exception of two samples which exceeded the Federal MCL of 0.05 mg/L for manganese. It should be noted that the water standard for manganese is established based on aesthetic concerns of taste, or odor; thus, exceedance of this standard does not constitute a health concern. These samples represented the Marion intake (0.09 mg/L) and the Refuge treated water (0.28 mg/L). As shown in Exhibit B, the Phase I metals analyses are questionable because of QA/QC deficiencies. The City Marion treated water contained concentrations of bromodichloromethane (17 ug/L) and chloroform (180 ug/L) which exceeded the Federal MCL of 100 ug/L for total

trihalomethanes for the Refuge Treated Water Supply slightly exceeded the Federal Drinking Water MCL of 100 ug/L. These compounds are likely formed as a result of chlorination. Corrective measures have been taken at the Refuge Water Treatment Plant to reduce levels of trihalomethanes. Acetone, methylene chloride and isophorone were also detected but were also present in the method blanks. Cyanide levels (0.09 mg/L but not detected in the duplicate) in the sample from the City of Marion treated supply were above the Illinois General Use and Public Water Supply standards, but below the lifetime health advisory of 154 ug/L for cyanide in drinking water.

Three lake waters contained low parts per trillion levels of PCBs. Locations 1B, 2C, and 6D (Samples 34-6, 7, 11) contained 0.008, 0.019 and 0.009 ug/L respectively; but only sample 34-7 at location 2C exceeded the chronic AWQC of 0.014 ug/L for protection of aquatic life. PCBs were not detected in any of the other lake water samples (detection limit 0.005 ug/L). (See Figure 38-2.) Traces of arsenic (2.0 to 3.4 ug/L), chromium (1 to 7 ug/L) and lead (1.2 to 9.2 ug/L) were detected, but only arsenic exceeded the human health AWQC of 0.0022 ug/L. The cyanide concentration (0.29 mg/L) exceeded the Illinois standards and the AWQC criterion in one water sample (Location 1B or No. 34-6); cyanide was undetected in all other samples at a detection level of 0.05 mg/L. Likewise, in one water sample (Location 10J, No. 34-15), the mercury concentration (0.4 ug/L) exceeded the ambient water quality standard for human health (0.144 ug/L). Sediment analyses indicated the presence of trace quantities of base/neutral/acid compounds. Di-n-butyl phthalate was detected in all ten sediments, ranging from 1,000 - 2,240 ug/kg wet weight; however, this compound was also detected in the QA/QC blanks.

CRAB ORCHARD LAKE
WATER COLUMN DATA



LEGEND

● SAMPLE SITE

PCB CONCENTRATIONS, ug/L

( )
( )
( )

MAY 28-29,1983 SURFACE WATER (PCBs, ug/L) (NTE, 1984)
MAY 21,1984 (NTE, 1984)
DECEMBER 11-16,1986 -PHASE II SAMPLING, WATER COMPOSITES (AT THREE DEPTHS)

O'BRIEN & GERE ENGINEERS INC

trihalomethanes. The formation of these compounds can be attributed to chlorination from the disinfection of drinking water supplies and is not indicative of off-site contamination sources. The City of Marion has instituted modifications to the treatment process and is now in full compliance for trihalomethanes, according to Illinois EPA's drinking water group. However, the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data. (See Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present.

38.3.2 Phase II Analytical Results:

The water samples collected from Crab Orchard Lake and from various raw and treated water supplies or potential water supplies were compared to the Illinois and Federal drinking water standards. Raw water supplies were also compared to the Ambient Water Quality criteria.

Two raw water supplies (Refuge Intake and Marion Reservoir Auxiliary Intake) contained trace amounts of methylene chloride (24 and 6 ug/L), barium (36 and 33 ug/L), and lead (4.8 and 8.5 ug/L). The detected concentrations for barium and lead are typical of raw surface water supplies, and are within the relevant standards. PCB concentrations were below the detection level of 0.005 ug/L in all of the current or potential drinking water samples. Methylene chloride, though detected, was also detected in the blank samples and may be a result of sample or laboratory handling. The treated water samples (Refuge treated water and City of Marion treated water) contained bromodichloromethane (11 and 4 ug/L respectively), and chloroform (94 and 35 ug/L). Total

Bis (2-ethylhexyl) phthalate was detected at 750 ug/kg in sample location #8F, No. 34-23. The lake control sample collected from the west end by the spillway dam (location #10J, No. 34-25), remote from any industrial areas of the Refuge, also contained phthalates (480 ug/kg wet weight), pyrene (450 ug/kg wet weight), and fluoranthene (510 ug/kg wet weight).

PCB concentrations were below the 40 ug/kg wet weight detection limit in all but two sediment samples (See Figure 38-1). Sample location #2C (No. 34-17) collected near the Area 9 Landfill area contained a PCB concentration of 248 ug/kg (291 ug/kg duplicate); while sample location #1B (No. 34-16) contained 104 ug/kg PCBs. Arsenic was detected in all ten sediments at concentrations ranging from 9.7 to 24 mg/kg, but duplicates and recoveries were outside control limits for these analyses. All other compounds analyzed were below the detection limits. The field data for the fish samples is presented as Table 38-1 and includes the weight and size for each fish sample collected. The analytical results for pesticides/PCBs, cadmium, mercury, and lead are presented in Table 38-2. Figure 38-3 shows the sampling locations and PCB results. Most parameters were below the detection levels, but concentrations of Aroclor 1254 ranged from below 0.4 to 6.4 mg/kg wet weight. Two carp samples, 34-29 and 34-30 from location #2C, contained 6.4 and 3.0 mg/kg PCBs respectively (3.9 and 4.4 mg/kg when re-analyzed); these concentrations are above the FDA action level of 2.0 mg/kg for PCBs. It should be noted that FWS conducted a split sample analysis for sample 34-29 and obtained a result of 1.2 mg/kg PCB wet weight, which is below the FDA action level, and is significantly below the earlier results for this sample. Fish sample 34-27 from location #1B contained 1.05 mg/kg wet weight of mercury. This concentration is only slightly above the FDA action level of 1.0 mg/kg.

TABLE 38-1

FIELD DATA FOR FISH COMPOSITE SAMPLES

SAMP No.	I.D.	LAB No.	DUP/SPKE FVS	SPECIES	WEIGHT LBS.	LENGTH INCHES	WEIGHT LBS.	LENGTH INCHES	WEIGHT LBS.	LENGTH INCHES	WEIGHT LBS.	LENGTH INCHES	WEIGHT LBS.	LENGTH INCHES
438	34- 26	19170		CARP	3	18- 3/8	3	18- 3/4	4- 1/2	21- 1/2	4- 1/8	21- 1/8	2	16
439	34- 27	19171		BASS	1- 1/2	13- 7/8	1- 1/4	13- 1/2	2- 3/8	16	3- 7/16	18	1- 3/16	13
440	34- 28	19172	DUP	BASS	2- 1/4	15- 1/2	3- 9/16	17- 1/2	2- 1/2	16	1- 7/16	13- 7/8	1- 1/8	12- 3/4
460	34- 48	19192		BULLHEAD	0- 1/4	8	0- 1/4	7- 7/8	0- 9/16	10- 1/4	0- 5/8	10- 3/8	0- 1/4	8
460	34-		FVS	BULLHEAD	0- [illegible]	9	0- 5/16	8- 7/8	0- 1/4	8- 1/4	6	9- 3/8	0- 1/4	8
461	34- 49	19193	DUP	BULLHEAD	0- 1/4	8- 1/2	7	9- 1/8	2	6- 5/8	0- 1/4	8	0- 1/4	6- 7/8
462	34- 50	19194		CATFISH	9	12	4- 3/4	22- 7/8						
441	34- 29	19173		CARP	2- 1/8	17- 1/4	3- 1/4	19	1-13/16	16- 1/8	2-13/16	18- 3/4	2- 1/2	18- 2/8
441	34-		FVS	CARP	1- 3/4	15- 1/8	1- 1/4	13- 7/8	1- 1/8	13- 3/4	1- 1/8	13- 3/8	1- 1/4	14
442	34- 30	19174	DUP	CARP	2- 5/8	18- 1/8	2- 3/4	18- 3/8	3- 3/4	19	2- 1/4	17- 1/2	1- 1/8	14- 3/8
443	34- 31	19175		BASS	1-13/16	15- 1/8	2- 7/16	17	1-13/16	14- 7/8	1- 1/2	13- 7/8	1	12- 3/8
463	34- 51	19195		BULLHEAD	0- 5/8	10- 3/4	0- 3/8	8- 3/4	0- 3/8	9- 1/4	0- 3/8	9- 1/8	0- 3/8	8- 7/8
464	34- 52	19196	DUP	BULLHEAD	0- 3/16	6- 1/2	0- 3/8	9	0- 1/4	8	0- 1/2	10	0- 3/8	9- 1/8
465	34- 53	19197		CATFISH	9	28- 7/8	8- 1/8	26	0- 7/16	21- 7/8	3- 9/16	20- 7/8	4- 3/4	23- 1/8
444	34- 32	19176		CARP	2- 1/4	16- 7/8	1- 3/16	14- 3/8	1- 5/8	15- 1/2	0-13/16	12- 1/8	1-13/16	15- 3/8
445	34- 33	19177	DUP	CARP	1- 7/16	14- 5/8	3- 1/8	19	1- 5/8	14- 5/8	1- 9/16	15- 1/4	1- 9/16	14
446	34- 34	19178		BASS	2- 7/8	17- 1/8	2- 1/4	16- 1/8	0- 5/16	11- 1/8	1	12- 3/4	0- 9/16	10- 1/4
	34-		FVS											
466	34- 54	19198		BULLHEAD	0- 5/16	8- 1/2	0- 5/16	8- 7/8	0- 1/4	8- 1/2	0- 1/2	9- 3/4	0- 1/2	10
467	34- 55	19199	DUP	BULLHEAD	0- 9/16	10- 1/8	0- 9/16	10- 3/8	0- 1/4	8- 1/8	0- 1/2	9- 1/2	0- 1/2	9- 7/8
447	34- 35	19179		CARP	3- 7/8	20- 1/8	1- 3/8	13- 3/8	2- 1/8	16- 1/4	2- 7/8	19	2- 3/8	16- 1/8
	34-		FVS	CARP	2- 1/8	15- 1/4	2	16- 1/8	2- 3/8	16- 3/4	1- 3/4	15- 1/4	2	15- 1/8
448	34- 36	19180		BASS	0- 3/4	11- 1/8	4- 7/8	20- 1/2	3- 1/2	18- 5/8	1- 3/16	13- 1/8	0- 7/8	11- 1/4
449	34- 37	19181	DUP	BASS	0-15/16	11- 5/8	1- 3/4	14- 1/2	1-13/16	14- 7/8	2	15- 1/2	2- 3/4	16- 7/8
472	34- 59	19200		BULLHEAD	0- 7/8	11- 1/2	0- 5/8	10- 1/2	0- 7/16	9- 1/4	0- 3/4	11- 1/4	0- 7/16	9- 1/4
473	34- 60	19201	DUP	BULLHEAD	0- 5/8	10- 5/8	0- 1/2	9- 3/8	0-13/16	11- 3/8	0- 3/8	8- 1/2	0- 1/2	9- 1/2
471	34- 61	19202		CATFISH	1- 3/4	13- 7/8	7- 1/2	26- 1/2	2- 3/4	20	5- 1/4	23- 1/2		
450	34- 38	19182		CARP	2- 7/8	17- 5/8	6	23- 1/4	2- 3/4	17- 5/8	5- 3/8	22- 3/4	2- 1/8	16- 1/2
451	34- 39	19183	DUP	CARP	2	6- 1/8	1- 7/8	16- 1/8	2-11/16	18- 1/8	2- 1/4	17	2- 1/2	17- 1/8
452	34- 40	19184		BASS	0- 3/4	11	0- 3/16	6- 7/8	1- 1/2	14				
452	34-		FVS	BASS	5- 1/16	20- 7/8	0- 5/8	10- 1/2	0-11/16	11	0- 3/4	11- 1/2		
453	34- 41	19185	DUP	BASS	1- 3/4	14- 7/8	1- 1/8	12- 1/2	3- 3/16	17- 1/2	2- 5/16	15- 1/2	1- 1/4	13- 5/8
468	34- 56	19203		BULLHEAD	0- 1/4	7- 3/4	0- 7/16	9	0- 1/4	8	0- 1/4	8- 1/8	0- 1/4	7- 3/8
469	34- 57	19204	DUP	BULLHEAD	0- 1/4	8	0- 5/16	9- 1/4	0- 1/4	7- 3/8	0- 1/2	10	0- 7/16	9- 7/8
470	34- 58	19205		CATFISH	2- 5/8	19- 1/8	10- 3/4	29- 3/4	4	22- 1/4				

TABLE 38-2 (p. 1 of 2)
Analytical Results for Fish (Edible Tissue)

Location:	LAKE 1B	LAKE 1B	LAKE 1B	LAKE 1B	LAKE 1B	LAKE 1B	LAKE 2C	LAKE 2C	LAKE 2C	LAKE 2C	LAKE 2C	LAKE 2C	LAKE 3B	LAKE 3B	LAKE 3B
Sample ID:	34-26	34-27	34-28	34-48	34-49	34-50	34-29*	34-30	34-31	34-51	34-52	34-53	34-32	34-33	34-34
Sample Type:	Carp	Bass	Bass	Bullhead	Bullhead	Catfish	Carp	Carp	Bass	Bullhead	Bullhead	Catfish	Carp	Carp	Carp
Sample Number:	438	439	440	460	461	462	441	442	443	463	464	465	444	445	446
PESTICIDES/PCB (ug/Kg wet wt)															
115 4,4'-DDD	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
116 4,4'-DDE	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
117 4,4'-DDT	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
118 Aldrin	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
119 Alpha-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
120 Arochlor-1016	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
121 Arochlor-1221	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
122 Arochlor-1232	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
123 Arochlor-1242	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
124 Arochlor-1248	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
125 Arochlor-1254	1100	< 400	480	420	< 400	1300	6400	3000	540	940	850	1700	840	990	450
126 Arochlor-1260	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400
127 Beta-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
128 Chlordane	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
129 Delta-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
130 Dieldrin	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
131 Endosulfan I	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
132 Endosulfan II	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
133 Endosulfan Sulfate	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
134 Endrin	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
136 Endrin Ketone	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
137 Gamma-BHC (Lindane)	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
138 Heptachlor	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
139 Heptachlor epoxide	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
140 Methoxychlor	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
141 Toxaphene	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400
METALS (mg/Kg wet wt)															
166 Cadmium	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
178 Lead	< 0.05	< 0.05	< 0.05	< 0.05	0.13	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05
184 Mercury	0.172	1.05	0.420	0.072	0.070	0.120	0.152	0.239	0.303	0.102	0.078	0.120	0.122	0.114	0.201

FDA ACTION LEVELS
PCB = 2 mg/Kg
Mercury = 1 mg/Kg

(*) FWS Split Sample No. 441 contained 1.2 mg/kg wet wt. PCBs.

TABLE 38-2 (p. 2 of 2)
Analytical Results for Fish (Edible Tissue)

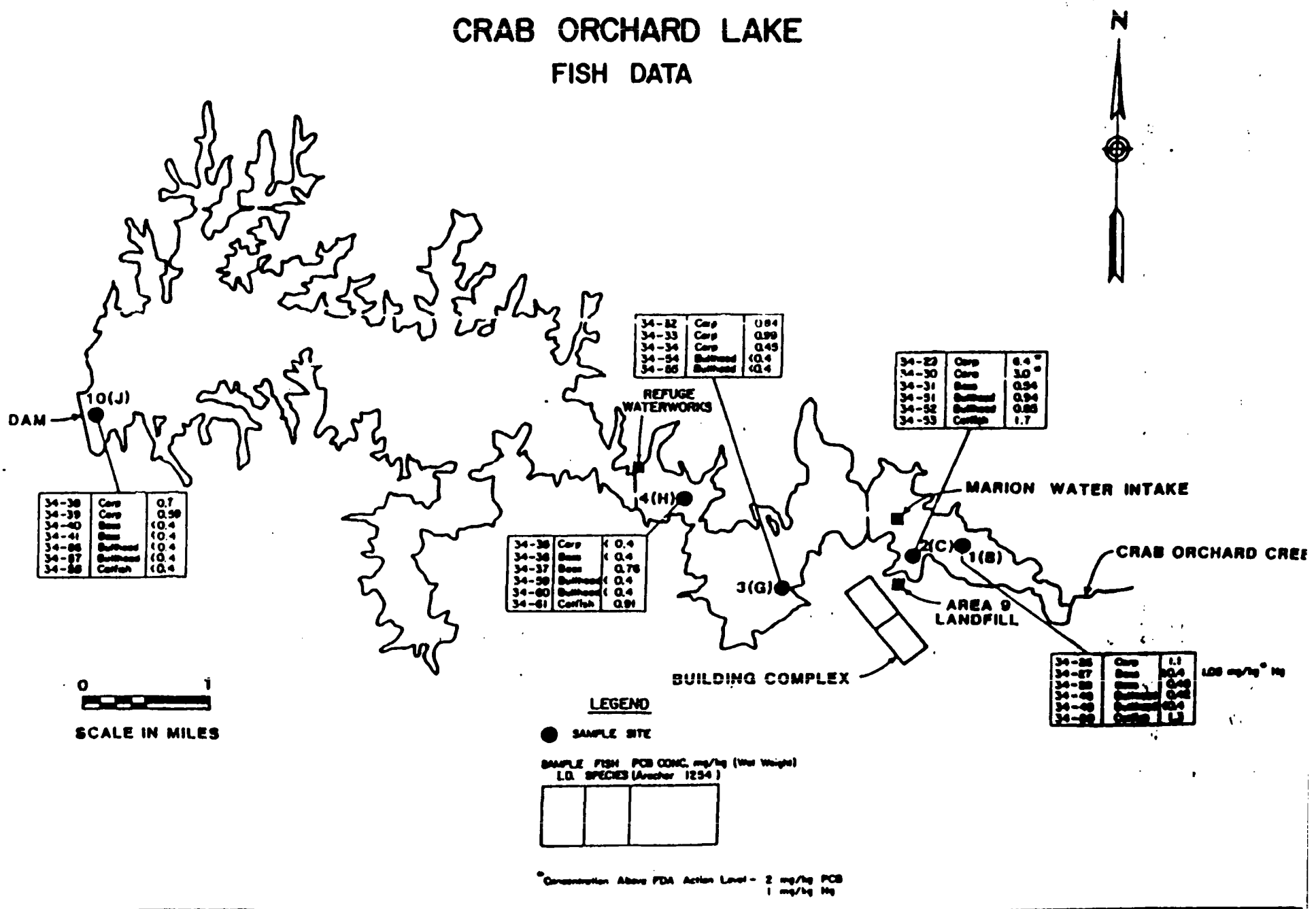
Location:	LAKE 3G	LAKE 3G	LAKE 4N	LAKE 4N	LAKE 4N	LAKE 4N	LAKE 4N	LAKE 4N	LAKE 10J	LAKE 10J	LAKE 10J	LAKE 10J	LAKE 10J	LAKE 10J	LAKE 10J
Sample ID:	34-54	34-55	34-35	34-36	34-37	34-59	34-60	34-61	34-38	34-39	34-40	34-41	34-56	34-57	34-58
Sample Type:	Bullhead	Bullhead	Carp	Bass	Bass	Bullhead	Bullhead	Catfish	Carp	Carp	Bass	Bass	Bullhead	Bullhead	Catfish
Sample Number:	466	467	447	448	449	472	473	471	450	451	452	453	468	469	470
PESTICIDES/PCB (ug/Kg we															
115 4,4'-DDD	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
116 4,4'-DDE	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	1800
117 4,4'-DDT	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
118 Aldrin	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
119 Alpha-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
120 Arechlor-1016	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
121 Arechlor-1221	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
122 Arechlor-1232	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
123 Arechlor-1242	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
124 Arechlor-1248	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
125 Arechlor-1254	< 400	< 400	< 400	< 400	760	< 400	< 400	910	700	590	< 400	< 400	< 400	< 400	< 400
126 Arechlor-1260	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400
127 Beta-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
128 Chlordane	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
129 Delta-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
130 Dieldrin	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
131 Endosulfan I	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
132 Endosulfan II	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
133 Endosulfan Sulfate	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
134 Endrin	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
136 Endrin Ketone	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
137 Gamma-BHC (Lindane)	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
138 Heptachlor	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
139 Heptachlor epoxide	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
140 Methoxychlor	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
141 Toxaphene	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400
METALS (mg/Kg wet wt)															
166 Cadmium	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
178 Lead	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	0.18	< 0.05	< 0.05
184 Mercury	0.078	0.052	0.116	0.094	0.217	0.055	0.061	0.055	0.091	0.079	0.057	0.108	0.034	0.063	0.168

FDA ACTION LEVELS
PCB = 2 mg/Kg
Mercury = 1 mg/Kg

(*) FWS Split Sample

CRAB ORCHARD LAKE

FISH DATA



O'BRIEN&GERE

Mercury concentrations in the remaining samples ranged from 0.034 to 0.420 mg/kg. Positive lead detections were found in two bullhead samples, 0.13 mg/kg wet weight in sample 34-49 from location #1B, and 0.18 mg/kg wet weight in sample 34-56 from the control location #10J. Cadmium was not detected at a detection limit of 0.10 mg/kg wet weight in any of the samples. There are currently no FDA action levels for cadmium or lead.

Kohler and Heidinger (undated) of Southern Illinois University (SIU) have recently reported preliminary results of a survey for PCB residues in Crab Orchard Lake fish. The SIU study was conducted independently of other residue surveys including the RI. The investigators collected largemouth bass, channel catfish, carp, bluegill, white crappie, and gizzard shad (a non-food fish) from three areas of the lake, including the embayment adjacent to the Area 9 Landfill (identified as Site 10 in the report). In an effort to discern age-related effects on accumulation of PCBs in these species, young, intermediate, and old individuals of each species (selected on the basis of life expectancy for each species) were analyzed. The results of this survey are presented in Section 2.7, and are summarized briefly below.

With the exception of channel catfish and possibly carp, no clear correlation of residue PCB level in fillet tissue with fish age was detectable in the study by SIU. This could be due to the limited number of samples in each data set (three fish per group) and the lower lipid content of species other than catfish, carp, and possibly shad (a major determinant in PCB accumulation with both species and fish age). These data are similar to the data developed during this RI in that they show that fish in the eastern portion of the lake have higher PCB concentrations than those

taken elsewhere in the lake. The results also demonstrate that carp and channel catfish contain higher residues than other species analyzed.

PCB analyses in the SIU study appear to have been conducted at a lower analytical detection limit, possibly 0.1 mg/kg, because a number of the pooled data presented are lower than the 0.4 mg/kg detection limit used in this RI. The composite data for most of the species and collection locations show residue levels in the range of 0.2 mg/kg PCBs. As discussed later, this provides a basis for treating RI fish data of less than the detection limit of 0.4 mg/kg as 0.2 mg/kg, one half the detection limit, when calculating averages for the quantitative assessment.

Due to the preliminary nature of the data presented in the SIU report, they are not used in the quantitative assessment of PCB-associated risks. However, the data are in general qualitative and quantitative agreement with the data collected during the RI and other investigations.

As discussed in the review of previous investigations in Section 2.7, Stuart (1984) collected Crab Orchard Lake fish and bottom sediments from the Area 9 embayment for a survey of polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). This survey did not detect the highly toxic 2,3,7,8-TCDD isomer in either fish or sediments. The 2,3,7,8-TCDF isomer was found in fish at an average level of 15.2 ppt (mean of 10 fish samples) with a maximum level observed of 41 ppt. In sediment, TCDF was detected at a mean of 19 ppt (for 5 samples) and a maximum of 50 ppt. Other isomers of PCDD were reported at ppt levels, but the isomeric distribution was not identified.

38.4 Environmental Effects

38.4.1 Qualitative Assessment

38.4.1.1 Source Evaluation

The western portion of Crab Orchard Lake is a focal point for recreational activities in the Refuge and supports an extensive ecological community as well. The lake has been used previously as an auxiliary drinking water source for the City of Marion. As discussed in Section 2.5, this auxiliary intake has been used one or two times within the last fifteen years, and is presently non-operational.

As a result of a variety of manufacturing activities along the eastern portion properties for four decades, the eastern part of the lake has served as a potential recipient of industrial discharges, landfill leachates, municipal wastewater and other possible pollutant sources. The results of chemical analyses of lake water, sediments, and biota, as described in the preceding section and in Section 2.7, suggest that the eastern part of the lake has in fact received industrial discharges.

For the purpose of this risk assessment, potential human health effects will focus on ingestion of finished drinking water drawn from Crab Orchard Lake and of fish taken from the lake by recreational angling. Because of the wide variety of subchronic and chronic effects documented in laboratory tests, often at low exposure levels, PCBs were chosen as an indicator contaminant for the risk assessment. The toxicological and physicochemical properties of PCBs are summarized in Exhibit A and in the quantitative assessment portion of this report.

Dibenzodioxin and furan residues in fish and sediments were analyzed in a survey conducted in 1984 (Stuart, et al., 1984) as

presented in Section 2.7. The risks associated with the detection of PCDD/PCDF residues in fish are evaluated in Section 38.4.2.2, Quantitative Risk Assessment.

38.4.1.2 Transport Route Evaluation

a) Air: Due to the nature of these compounds, PCBs do not volatilize from water to a significant degree at the low concentrations found in Crab Orchard Lake. The air route is therefore considered non-functional.

b) Direct Contact: PCB residues have been detected in lake sediments. The direct contact exposure route is thus functional but will be considered in this analysis in the assessment of ingestion exposures.

c) Surface Water: PCB residues have been detected in some water samples from the lake, but not in drinking water prepared from it or in the raw water intakes to the treatment plant. Thus, the surface water route is functional for both humans and wildlife. This exposure path will be considered as a form of ingestion exposure in the following assessment.

d) Ingestion: PCB residues have been detected in aquatic biota taken from the lake (see also Previous Studies in Section 2.7). Therefore, the ingestion exposure pathway is complete for both humans and wildlife.

38.4.1.3 Receptor Evaluation

Human

Use of Crab Orchard Lake as a source of drinking water is very limited, serving only the Refuge and the Marion penitentiary at

present. Potential receptors therefore include visitors and employees of the Refuge as well as residents of the penitentiary. Analyses of Refuge finished drinking water failed to reveal the presence of PCBs. Therefore, the only potential human receptors for PCBs in the lake are consumers of fish captured by recreational angling. A hypothetical case is also evaluated where the backup supply for the City of Marion might be used to replenish the Marion Reservoir in an emergency.

Wildlife

Exposures of fish and wildlife to PCBs are complete for the direct contact/exposure route. Benthic and bottom-feeding organisms in the lake will receive the highest exposure. Exposure to PCBs is also possible for wildlife feeding on contaminant residues.

38.4.2 Quantitative Risk Assessment

38.4.2.1 Estimates of Surface Water Exposures

Previous surveys of Crab Orchard Lake for surface water residues of PCBs and other contaminants were presented in Section 2.7.2. With the exception of an elevated level of PCBs in a bay adjacent to the Area 9 Landfill (0.16 ug/L, Hite, 1984 and IDPH, 1976-1987) all water samples from previous surveys were below the detection limit of 0.1 ug/L. The Phase II investigation found only 0.009 to 0.019 ug/L PCBs in the Area 9 bay water and no PCBs were detected elsewhere at a detection limit limit of 0.005 ug/L PCB.

It is conceivable that Crab Orchard Lake could be used in the future as an emergency backup supply of water for the City of Marion to supplement the Marion Reservoir. This has occurred as

recently as 1981, but the alternate water source has since been switched to Herrin Lake. During the last withdrawal from Crab Orchard Lake in 1981, about 6 percent of the capacity of Marion Reservoir was replenished. To determine the health risks associated with such a use of Crab Orchard Lake, it will be estimated that such an event occurs every ten years, and that the replacement water contains 9 ng/l (ppt) of PCBs, based on a sampling of lake water closest to the auxiliary intake.

The resultant PCB concentration from replenishing 6 percent of Marion reservoir's capacity would be 0.5 ng/l, a level which is 10 times lower than the detection limit for PCB analyses in water. If such a diluted concentration were maintained by periodic replenishment from Crab Orchard Lake, a chronic surface water ingestion scenario would exist. The main concern for health effects from chronic exposure to PCBs is potential carcinogenicity based on controlled laboratory studies. In order to assess risks associated with such exposure, daily PCB exposure rates are estimated and integrated with a measure of the potency of PCBs to produce human cancer as extrapolated from animal response data.

Assuming a standard lifetime (70 years) drinking water ingestion rate of 2 liters per day containing 0.5 ng PCB per liter, a daily PCB exposure for a 70 kg adult of 0.014 ng/kg/day is estimated. Using the PCB cancer unit risk factor of 7.7 $(mg/kg/day)^{-1}$ (Exhibit A), an estimate of excess cancer from this exposure route of 1.0×10^{-7} is obtained, under worst case conditions. This cancer risk is well below the 10^{-6} to 10^{-5} range of risk generally considered acceptable by regulatory agencies.

38.4.2.2 Estimates of Direct Contact Exposures

As discussed in Section 2.7.1, a 1983 survey (Hite, 1984) of lake sediments showed PCB levels of less than 0.01 to 0.27 mg/kg in a region next to the Area 9 Landfill. Other surveys (Ruelle and Adams, 1984) detected 0.41 and 0.76 mg/kg in the same bay region. Sediments from mid-lake north of the landfill, were below 0.05 mg/kg in the latter survey. Elevated PCB levels, presenting the opportunity for ingestion exposures by benthic organisms, are considered in the ingestion exposure discussion below. It is doubtful that PCBs bound to sediment are subject to dermal absorption by aquatic organisms but they might contribute to localized areas of elevated surface water concentrations which may be available for absorption.

38.4.2.3 Estimates of Ingestion Exposures

Human

As part of the field investigation, specimens of fish including bass, carp, bullheads, and catfish were collected from Crab Orchard Lake. The edible tissues of these species were analyzed for PCBs, pesticides, cadmium, lead, and mercury. The results, presented in Table 38-2 and Figure 38-3, indicate that the ingestion exposure route is complete for humans and wildlife consuming fish taken from certain portions of the lake. PCB residues were elevated primarily in sampling location #2C, with lower levels detected in locations #1B and #3C.

Only two of the samples (out of 29 analyzed) contained residues in excess of the FDA action level of 2 mg/kg. Both of these samples

consisted of carp taken from location #2C, near a general area previously found to contain PCB residues in lake sediments. Carp are a bottom-feeding species which would come into direct contact with contaminated sediments, as would other bottom feeders such as catfish. With the exception of the fish sampled at the control location #10J at the Crab Orchard Lake dam, all fish sampled were from portions of the lake which are not readily accessible to recreational fishing, and from which movement into the remainder of the lake is constrained by two rather narrow passages. Since neither bass, carp, bullheads, nor catfish are particularly migratory, it is unlikely that fish in the eastern end of the lake make a significant contribution to populations in the remaining portion of the lake.

This ingestion exposure risk analysis involves a 'worst case' conservative estimate based on an individual who relies upon fish from Crab Orchard Lake for his or her total fish diet over a 5, 10 or 70 year period. The risk levels presented in Table 38-3 assume that PCB residues in fish remain constant over the specified period of time. Each of these assumptions may be improbable. For instance, it is more likely that an individual would obtain some portion of his fish diet from other sources, including ocean fish, fish from nearby lakes or rivers, or fish from commercial fisheries, which would diminish the risk estimates proportionately. In addition, PCB residues in fish from Crab Orchard Lake are likely to decline over time, because of decreasing levels available for uptake from bottom sediments due to the natural sedimentation process in the lake and remediation of on-land potential contaminant sources, and continuing replacement of aquatic populations.

It is apparent from examining the range of risks presented in Table 38-3 for ingestion of fish under various dietary scenarios that the most sensitive variable in the risk estimate is the total quantity of fish captured from Crab Orchard Lake and consumed over a lifetime which contains the level of PCB fillet residues currently observed. The upper bound cancer risk estimate of 2.0×10^{-3} represents the consumption of approximately 30 g of catfish (which are bottom feeders) from Crab Orchard Lake every day during a 70 year lifetime (sports fisherman scenarios, Table 38-3). This estimate assumes that 95 percent of the catfish are taken from the western area of Crab Orchard Lake, and 5 percent are taken from the east, since the fishable acres in the east end comprise about 3 percent of the total 7,000 fishable acres in Crab Orchard Lake. In addition, due to its average water depth of 2-3 ft., the east area is inaccessible to boat fishing and therefore fewer fish would be taken from the east side of the lake. More realistic consumption scenarios for persons who obtain their entire fish diet solely from Crab Orchard Lake (if any) may be those risk levels derived for periods of 5 and 10 years, with associated risks between 10^{-5} and 10^{-4} (one in one hundred thousand to one in ten thousand).

Cancer risks can also be presented in terms of the number of meals of fish from Crab Orchard Lake. For individuals consuming fish from the western portion of the lake, about 40 to 80 meals of mixed fish, assuming an average meal size of 0.5 to 0.25 lb and a catch similar to the creel census, would result in a lifetime excess cancer risk of about 10^{-5} (one in a hundred thousand). Therefore, if an individual wished to limit their lifetime excess cancer risk to one

TABLE 3B-3
CRAB ORCHARD LAKE

ESTIMATED RISK TO HUMANS
DUE TO CONSUMPTION OF FISH TISSUE

ASSUMPTIONS:
- 100 % of fish diet is captured at Crab Orchard Lake.
- Consumption of Crab Orchard fish continues over a 70-year lifetime, or during a 10-year or 5-year period.
- Undetected values are calculated as one half the analytical detection limit (0.2 mg/kg for RI data).
- Cancer unit risk factor of 7.7 $(mg/kg/day)^{-1}$ for Aroclor 1260 (*)
- East/West division denoted by Wolf Creek (200 fishable acres on eastern area, 7000 fishable acres total for lake.)

SCENARIO	AVERAGE PCB CONCENTRATION (mg/kg ww) (1)	(2)	<------ RISK LEVELS+ ------> 70-year Lifetime	10-year Exposure	5-year Exposure
I. Average Fisherman (6.5 g fish per day or 10-20 meals per year)					
a) 95% West Bass/5% East Bass	0.34	0.20	2.5E-04	3.5E-05	1.8E-05
b) 100% West Catfish/Bullhead	0.32	0.08	2.3E-04	3.3E-05	1.6E-05
c) 95% West /5% East Catfish	0.61	0.51	4.3E-04	6.1E-05	3.0E-05
d) Creel Census (4)	0.30	--	1.7E-04	2.4E-05	1.2E-05
e) National Average (5,#) 1976-1979	0.29	--	2.1E-04	3.0E-05	1.5E-05
f) National Average (6,#) 1980-1981	0.18	--	1.3E-04	1.8E-05	9.2E-06
II. Sports Fisherman (30 g fish per day or 50-100 meals per year)					
a) 95% West Bass/5% East Bass	0.34	0.20	1.2E-03	1.6E-04	8.3E-05
b) 100% West Catfish/Bullhead	0.32	0.08	1.1E-03	1.5E-04	7.4E-05
c) 95% West /5% East Catfish	0.61	0.51	2.0E-03	2.8E-04	1.4E-04
d) Creel Census (4)	0.30	--	7.8E-04	1.1E-04	5.6E-05
e) National Average (5,#) 1976-1979	0.29	--	9.6E-04	1.4E-04	6.8E-05
f) National Average (6,#) 1980-1981	0.18	--	5.9E-04	8.5E-05	4.2E-05

NOTES & REFERENCES:

(1) Averages are calculated assuming fish without detected PCB residues contain such residues at one half the analytical detection limit.
(2) Averages are calculated assuming fish without detected PCB residues are free of such residues.
(3) Derived using a 1976 Creel Census survey and average concentrations in fish species detected in the RI and in monitoring studies conducted by the State of Illinois (see Section 2.7). Based on the Creel Census data, the relative catch per boat expedition at Crab Orchard Lake is comprised of roughly, 35% bass, 31% bluegill sunfish, 14% catfish, 12% crappie and 8% bullhead.
(4) ATSDR (November, 1987). Draft Toxicological Profile on PCBs.
(5) Schmidt, CJ et al. (1985). National Pesticide Monitoring Program. Arch. Environ. Contam. Toxicol.; 14:225-60.

(#) Fillet residues calculated as one third reported whole body residue.
(*) The potency factor of 7.7 $(mg/kg/day)^{-1}$ is based on studies using Aroclor 1260; only Aroclor 1254 residues were detected at Crab Orchard Lake. Available data neither demonstrate nor preclude the carcinogenicity of Aroclor 1254.
(+) Additive risks due to PCB/TCDF residues in fish might be obtained by adding 15 percent to risk level noted for PCBs.

chance in one hundred thousand, the total number of meals consumed in a 70-year lifetime from Crab Orchard Lake (western side excluding large channel cat) should be in the range of 40 to 80 meals. From a practical standpoint, four to eight meals annually over a ten year period would result in acceptable levels of risk similar to other dietary risks. For a person only occasionally visiting the lake, a higher rate of fish consumption would not yield unacceptable risks.

Stuart et al. (1984) presented the results of analyses of Crab Orchard Lake fish for polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). These were whole fish samples of largemouth bass, channel catfish, and carp captured from the eastern portion of the lake. The most highly toxic of this class of chemicals, 2,3,7,8-tetrachlorodibenzodioxin (TCDD), was not detected. One catfish contained unspecified penta- and hexachlorinated PCDD. Some of the fish analyzed contained 2,3,7,8-tetrachlorodibenzofuran (TCDF) at a mean level of 15 ppt, and two fish of the ten analyzed contained other TCDF isomers, with a group average of 2.5 ppt. Clark (1988) has indicated that in the environment, the 2,3,7,8-TCDD and TCDF compounds, the more toxic congeners of these classes, appear to preferentially bioaccumulate. This might in part explain why other TCDF congeners present in commercial PCBs were not detected in this fish survey.

Fink (1986) uses a conversion factor of 1/3 to estimate fillet concentrations of bioaccumulated organic contaminants to fillet concentrations. Therefore, the whole body TCDF data are converted to 5 ppt of 2,3,7,8-TCDF and 0.8 ppt of other TCDFs in fillets. Under the upper bound fish consumption scenario constructed for

PCBs, and assuming lifetime daily consumption of 30 grams catfish with no difference in PCDF concentration between the east and west portions of the lake, a daily adult (70 kg) intake of 2.1×10^{-9} mg/kg/day of the 2,3,7,8-TCDF and 0.36×10^{-9} mg/kg/day of other TCDFs is estimated.

In the absence of an extensive toxicological data base for PCDFs in general, U.S. EPA (1987, 1988) has adopted a weight of evidence approach to risk assessment of these compounds, termed the toxic equivalency factor approach, as an interim risk assessment procedure. In this approach, available test data and quantitative structure-toxicity relationships have been reviewed in order to rank the relative toxicity of the various PCDD and PCDF compounds and congeners relative to the best characterized and most toxic compound of these groups, 2,3,7,8-TCDD. Under this approach, the 2,3,7,8-TCDF congener is given a toxicity equivalence factor of 0.1 relative to 2,3,7,8-TCDD, and all other TCDFs are given a factor of 0.001. The cancer potency factor of 2,3,7,8-TCDD is currently given as 1.56×10^{5} $(mg/kg/day)^{-1}$. Therefore, the potency factors of 2,3,7,8-TCDF and other TCDFs are estimated as 1.56×10^{4} (mg/kg/day) and 1.56×10^{2} $(mg/kg/day)^{-1}$, respectively. The lifetime cancer risks from exposure to these compounds at the above estimates of exposure are, consequently, 3.3×10^{-3} for 2,3,7,8-TCDF and 5.6×10^{-7} for the other TCDFs. Since the risk estimate for PCB exposure at the same fish consumption rate is 2×10^{-3}, the TCDF exposure might increase overall risk to 2.3×10^{-3}, a factor of 15 percent. This incremental risk factor could be added to all risk

estimates presented in Table 38-3 to give an upper bound additive risk for PCB and TCDF exposure.

In view of the inherent uncertainty in quantitative carcinogenic risk assessment, the biological significance of this estimated incremental risk is unclear. In addition, some of the PCDD-like congeners present in aroclor mixtures often used in toxicological evaluations of PCBs may be responsible for some portion of the observed toxicity (Hileman, 1988; U.S. EPA, 1987), depending on the effect studied and the biochemical mechanism involved. If this applies to this risk assessment, the incremental risk from TCDF exposure via fish consumption may be somewhat overstated. However, in view of the possibility that certain PCDD and PCDF congeners may preferentially accumulate in sediments and in fish, including the more toxic 2,3,7,8-tetrachloro derivatives, environmental mixtures might be enriched in these components relative to commercial mixtures. Therefore, a conservative approach to this aspect of the risk assessment is desirable.

Short term consumption of catfish containing the average PCB levels (Scenario C. of Table 38-3) observed at Crab Orchard Lake does not result in intake levels which would be a cause for concern from non-carcinogenic PCB toxicity. Using a threshold exposure level for short-term non-carcinogenic effects of 1 mg/kg/day and a safety factor of 100, U.S. EPA (1986) established a short term acceptable intake level of 0.7 mg/day for a 70 kg adult for a ten day exposure period. At a fish ingestion rate of 250 g/day, assuming one meal of catfish daily during ten consecutive days, and a fishing pattern of 95 percent of catfish obtained from the west end and 5 percent of catfish

from the east end of Crab Orchard Lake, a daily PCB intake of 0.61 mg/kg x 0.25 kg/day or 0.15 mg/day is estimated. This intake is well below the acceptable level of 0.7 mg/day for short term exposures.

For longer term exposures, ATSDR (1987) and U.S. EPA ECAO-CIN-414 (1987) have identified a no observed adverse effect level (NOAEL) for Arocior 1016 for developmental toxicity (reduced birth weight) of 0.01 mg/kg/day upon chronic oral exposure in monkeys. U.S. EPA ECAO-CIN-414 (1987) established a long-term health advisory Reference Dose for non-carcinogenic effects by applying a 100-fold uncertainty (safety) factor to this NOAEL, or 0.0001 mg/kg/day. At an ingestion rate of 30 g fish per day x 0.61 mg PCB/kg fish, a daily exposure rate of 0.00026 mg/kg/day for a 70 kg woman is estimated. This upper bound exposure estimate of 0.00026 mg/kg/day is higher than the Health Advisory intake level. However, this exposure level does not necessarily imply that developmental toxicity will be observed in humans exposed at this level because of the conservative nature of the assumptions and safety factors used. In fact, this exposure level is 50 percent lower than the lowest observed adverse effect level (0.0004 mg/kg/day LOAEL) seen in the monkey study with a 100-fold uncertainty factor applied. In addition, the rhesus monkeys studied were more sensitive to PCB effects than another highly sensitive species, such as the mink.

To place these estimates in perspective, and to illustrate the sensitivity of the risk analyses to the average concentrations detected in fish (as well as to the procedure for analyses of fish samples), the risk levels associated with consumption of freshwater fish available in

interstate commerce in the U.S. were reviewed. The U.S. Fish and Wildlife Service reported that whole body PCB residues in freshwater fish nationwide in 1980-81 averaged 0.53 ppm (Schmidt et al., 1985). As an approximation, the edible portions of fish tissue are often assumed to contain one third the lipophilic contaminant level of whole fish due to decreased fat content (Fink, 1986). Thus, under the same conservative dietary assumptions used in this assessment (30 g of fish consumed every day over a 70-year lifetime) for an avid freshwater sport fisherman, it is estimated that a fillet PCB level of 0.53/3 or 0.18 mg/kg - the average level throughout the country in 1980-1981 - has an associated risk of 5.8×10^{-4}.

Similarly, the estimated risk level associated with consumption of freshwater fish containing the national average concentrations in edible tissue reported for the period between 1976 and 1979 is 9.6×10^{-4}, assuming the same lifetime consumption rate of 30 g fish daily (see Table 38-3). This risk estimate is proportional to the higher national average PCB concentration reported in ATSDR (1987) for the period between 1976 and 1979. The downward trend in residue content would appear directly related to decreases in general environmental PCB residues due to restrictions on the manufacture and use of PCBs. Risks to the average consumer of fish would be substantially less, as saltwater fish, containing much lower PCB residues than freshwater fish, comprise a significant portion of fish consumed in the typical diet.

Background risk levels are largely a function of the analytical protocol (and analytical detection limit) employed in the analysis of fish samples. This dependence is illustrated, in part, by the

magnitude of the above estimated risks associated with consumption of commercially available fish containing the nationwide average PCB concentration reported in ATSDR (1987). To illustrate the direct dependence of the risk levels estimated for this site to the detection limit of 0.4 mg/kg employed in this analytical program (a common detection limit utilized nationwide in the analysis of PCBs in fish tissue), the risks associated with consumption of only fish flesh with results below the detection level were estimated. Assuming the undetected results are averaged as being equal to the detection limit of 0.4 mg/kg, and are consumed at a rate of 30 g daily during a 70 year lifetime, the risk is estimated to be 1.3×10^{-3}. Thus, in order to provide a less biased while still conservative estimate, scientists and regulatory personnel often utilize one half the analytical detection limit as a proxy for the undetected results in computing the average of a particular data set. Using this procedure to re-evaluate the risks associated with consumption of fish samples which do not contain detectable PCB levels results in a risk level of 6.6×10^{-4}. Use of the preliminary data reported in recent studies by SIU (Kohler, undated), which revealed an average PCB concentrations of 0.2 mg/kg in fish tissue from Crab Orchard Lake, would yield comparable risk values as using one half of the detection limit of 0.4 mg/kg.

There are several factors in addition to the total quantity of fish consumed, the estimated average level of contamination, and the analytical detection limit which may have a direct bearing upon the nature of this analysis. These factors should also be considered when evaluating and using the above risk estimates:

1. It is well recognized that certain methods of cleaning and cooking fish can remove significant amounts of contaminants found in fatty portions of edible fish tissue, including PCBs (Cordle et al., 1982). For instance, mean whole body PCB residues in lake trout from Lake Michigan in 1973-1974 of 18.9 to 22.9 mg/kg were decreased to 1.03 - 4.67 mg/kg after cleaning (i.e. removing head and internal organs) and cooking. Zabik et al. (1979) also demonstrated a significant reduction of PCB levels in lake trout fillets containing large amounts of fat by various cooking techniques including broiling, roasting, and microwaving. The degree of residue removal was dependent on the cooking method, with broiling and roasting more effective than microwave preparation.

Skea et al. (1979) showed a lower degree of removal using less fatty fish; brown trout and small mouth bass. Smoking removed 12 percent of Arocior 1254 residues in the brown trout, whereas broiling increased the residue level, presumably by removing tissue water. In the smallmouth bass, baking increased the residue level while deep frying reduced the level by nearly 50 percent. Zabik et al. (1982) compared several preparation methods for effects on PCB residue level in cooked and raw carp fillets and reported either no effect or an increase in concentration. Thus, the degree of residue reduction of PCBs by cooking seems dependent on the species concerned, primarily due to variation in fat content, and cooking methods used, due to varying efficiencies for removal of fat and tissue water.

The presented assessment assumes that all PCBs in the fillet before cooking are carried over in the cooked meal. It is quite possible, however, that some portion of the residues would be lost during preparation of the food by certain methods, although no experimental data were located for the species assessed (largemouth bass and catfish). This lack of specific data and the variability observed in the literature precludes a quantitative estimate of possible risk reduction by cooking. As a result, further investigation on the effect of preparation of these species would be a useful aid for further refinement of this site assessment.

2. As a conservative precaution, carcinogenic risks were estimated as if the PCBs in question were of a potency equivalent to Aroclor 1260, whereas only Aroclor 1254 residues have been observed in Crab Orchard Lake sediments and fish. Under the conditions of a National Cancer Institute (NCI) chronic bioassay of two-year duration, Aroclor 1254 did not demonstrate convincing evidence of carcinogenicity in rats (as reviewed in Exhibit C, Harbison et al., 1987). Because the experimental design of the NCI study may have lessened the power to detect elevated tumor incidence, U.S. EPA has assessed the risks of all PCB aroclors including 1254 as if these were all of equivalent potency to Aroclor 1260. Two studies of greater than two-year duration in rats with Aroclor 1260 produced elevated rates of liver tumors in the late stages of the animals' lives. As reviewed in U.S. EPA ECAO-CIN-414 (1987) and ATSDR (1987), the

Aroclor 1260 studies have been selected by regulatory agencies as the basis for carcinogenic risk assessment of all PCBs.

Based upon what is known about the comparative biochemical toxicology of PCB congeners, however, Aroclor 1254 may be less potent as a carcinogen than Aroclor 1260. The more heavily chlorinated PCBs (penta- and hexachlor-) are believed to be most toxic to the liver, and these congeners predominate in Aroclor 1260 relative to 1254 (ATSDR, 1987). Thus, even if Aroclor 1254 were to show carcinogenic properties in rats if tested under the same conditions as Aroclor 1260, it is possible that the effect would be less, due to lesser presence of highly chlorinated organics.

This argument concerning the potentially lesser potency of Aroclor 1254 may not be applicable to environmental exposures in all cases. Because of their relatively low water solubility and greater persistence, environmental Aroclor 1254 residues in sediments and fish may be enriched in heavily chlorinated congeners relative to the commercial mixture as manufactured. Thus, the environmental mixture of Aroclor 1254 congeners might on occasion more closely resemble toxicologically a commercial Aroclor 1260 mixture compared to a commercial 1254 mixture. This does not, however, appear to be the case for the fish residue data generated in the RI. Examination of the gas chromatograph scans by a PCB analytical specialist indicates that the RI Aroclor 1254 patterns were representative of standard Aroclor 1254, with little of no enrichment of the more heavily chlorinated congeners (Hill, 1988).

The study by Norback and Weltman (1985) which forms the basis for the quantitative potency estimate for Arocior 1260 and all other PCBs, is somewhat flawed and its relevance to human risk assessment has been questioned (Exhibit C, Harbison et al., 1987). The dose administered appears to have been hepatotoxic, excessive control mortality was observed, and a number of the animals which were scored as tumor-bearing had received a partial hepatectomy (a possible tumor promoting influence) earlier in the study. The liver tumors induced in treated animals appeared very late in the animals' lives, and did not appear to be life-threatening or metastatic. These observations inject additional uncertainty into the quantitative assessment of the health risks from ingestion of fish containing PCBs. Use of these data helps ensure, however, that the assessment of risks associated with PCB exposure errs on the side of overstating rather than understating the risks.

The factors discussed above, as well as those presented elsewhere in this report (Section 6.5 and Section 38.4.3, Analysis of Uncertainties), illustrate the necessity to consider the underlying uncertainty as well as the overall weight of evidence when applying this approach to a case-specific assessment.

Humans might also receive exposures to PCBs from consumption of certain duck and geese species which may have been exposed to residues from bottom sediments in the eastern area of Crab Orchard Lake. According to the Refuge Manager, the 43,000-acre Refuge supports a peak population of approximately 12,000 ducks and 100,000 geese during the fall and winter seasons. The duck population

consists of about 10,000 puddle ducks (surface feeders) and 2,000 diving duck species which spend an average of two months/year at the Refuge. Both geese and ducks utilize the eastern end of the lake for resting and loafing purposes. These species generally feed very little in the lake because farm crops are abundant throughout the Refuge. Diving ducks, although seen occasionally resting in the east end, generally prefer deeper water areas in the western portion of the lake. Although the number of waterfowl on the eastern end mudflats or shallows may reach 2-4,000 birds for several days, individual birds and subflocks are constantly changing with other groups elsewhere on the Refuge on a daily or hourly basis. Thus, an individual bird is estimated to spend no more than 10 days maximum per season in the eastern portion of Crab Orchard Lake. Based on this limited exposure to sediments in the east area of the lake, and the low residue levels present, ducks would not be expected to accumulate significant PCB residues. The ingestion of duck or geese from Crab Orchard Lake is not considered to represent a risk to humans or wildlife at the Refuge.

In summary, the estimated risks to humans associated with consumption of fish are within a range of 10^{-5} excess lifetime cancer risk for rates of fish consumption between 40 and 80 meals occurring over a few years or over a lifetime of 70 years. At higher rates of consumption (greater than a meal per month or 12 meals/year), for more than three consecutive years or a lifetime, excess cancer risks would exceed 10^{-5} for most consumption scenarios.

Wildlife

The presence of PCB residues in fish in Crab Orchard Lake creates a complete exposure pathway for piscivorous birds and mammals in this region. Mink, otters, and other fish-consuming mammals living near the eastern portion of the lake may be exposed to the levels of PCB residues which have been detected in sediments and bottom-feeding fish. In particular, mink are highly susceptible to the acute and subchronic effects of PCBs (Newell et al. 1987), and thus will be a focus of the wildlife risk assessment. Piscivorous and/or aquatic birds may also be exposed to PCB residues since such birds would be expected to obtain a portion of their total diet from captured fish or fish carrion taken from Crab Orchard Lake. As discussed below, the Refuge is an active nesting area for 2-3 pairs of bald eagles, an endangered species with documented sensitivity to chlorinated hydrocarbon pesticides and perhaps PCBs. Herons and osprey may also be exposed via ingestion of fish obtained from some portions of Crab Orchard Lake. Certain species of duck might obtain part of their diet from Refuge fish; however, according to the Refuge Manager, such species spend approximately two months per year at the Refuge as a temporary habitat during their excursion north, and, during this period, spend most of their time feeding or loafing in the deeper areas of the lake, on the west end of the Refuge. Less migratory species of duck and/or geese which might remain at the Refuge long enough to receive limited exposure to contaminants are predominantly vegetarian, and tend to feed on various agricultural crops surrounding Crab Orchard Lake. For the above reasons, duck and geese will not be considered in the quantitative assessment.

In order to assess risks to piscivorous mammals, it will be assumed that a population of mink exists on the eastern portion of Crab Orchard Lake. As reviewed in Newell et al. (1987), a typical mink diet consists of about 50 percent fish, and a 1 kg adult mink consumes approximately 150 g of food per day, or 75 g of fish. Fish in the eastern portion of the lake (17 samples, all species) contained an average of 1.2 mg/kg in the muscle fillet ("edible tissue"). Whole body residues may be reasonably assumed to be three times as high (Fink, 1986), or 3.6 mg/kg. Thus, a mink consuming 75 g per day of fish containing 3.6 mg/kg of PCBs would receive a daily exposure of 0.27 mg/kg body weight/day. Newell et al. (1987) estimated that a daily intake of 0.13 mg/day would provide a margin of safety of 5 fold against the lowest level of dietary PCB exposure associated with reproductive impairment in laboratory tests with mink. Thus, the worst case exposure scenario provides a safety factor of more than 3 fold over the lowest observed effect level in mink. In view of the observation that mink tend to feed predominantly on young fish which have not had the opportunity to acquire elevated contaminant levels (Ruelle, 1987), this safety factor is concluded to be adequate. Piscivorous mammals less sensitive than mink would be expected to be further below the effect threshold, because mink are the species with the greatest documented sensitivity to PCBs.

In contrast to a resident mink population, piscivorous birds on the Refuge can obtain food from a number of sources, including all of Crab Orchard Lake and other aquatic systems. In developing criteria for contaminants in fish flesh in regard to piscivorous wildlife, Newell et al. (1987) utilized a food consumption rate of 20 percent of body

weight per day for a number of piscivorous birds, including bald eagle, great blue heron, and osprey. An adult bald eagle body weight of 4.5 kg is used in the assessment below. According to the Refuge Manager, the peak winter population of eagles on the Refuge is 30 birds. If these winter eagles restricted their food source to the vicinity of Crab Orchard Lake, their diet would consist almost entirely of wounded waterfowl and miscellaneous road kills such as snakes, squirrels, and rabbits. During the spring and summer (May through August) only 2-3 nesting pairs of bald eagles utilize Crab Orchard Lake; the diet of these birds during these four months is made up of approximately 80 percent fish species and 20 per cent carrion. It is reasonable to assume that the total fish diet is taken randomly from all areas of the lake, given the large foraging range of this species. If sampling locations #1B, #2C, and #3G in Figure 38-3 are considered to be representative of the eastern area of the lake (east of Wolf Creek), and sampling locations #4H and #10J are considered representative of the species on the west side of the lake, the average fillet tissue concentrations of PCBs for all fish species from the west and east portions of the lake are 0.37 and 1.2 mg/kg respectively. (Similar to the assumptions for the human health risk assessment, samples with results below the detection level are considered as one half the analytical detection limit or 0.2 mg/kg). The relative volume of water in the west portion of the lake is greater, since the average depth of the western area is 8-9 ft. versus 2-3 ft. on the eastern side. In addition, the west area represents approximately 97 percent of the total fishable acres in Crab Orchard Lake. If it is assumed that the fish population is

roughly proportional to the area of the lake (3 percent east, 97 percent west), then the weighted average PCB fillet tissue concentration would be 0.39 mg/kg for the overall fish population in Crab Orchard Lake. The whole body concentrations consumed by eagles would be estimated to be three times this concentration (Fink, 1986) or 1.2 mg/kg.

Considering the above assumptions, a 4.5 kg adult eagle is estimated to consume 0.19 mg PCBs/kg/day during the four summer months when its intake is comprised of 80 percent whole fish and it maintains a consumption rate of 20 percent its body weight. Normalizing this intake over a one year period to be representative of both winter and summer dietary intakes, an exposure level of 0.057 mg/kg/day is estimated for eagles. Applying a 10-fold margin of safety to a no observed effect level (NOEL) for reproductive effects in PCB- exposed domestic hens (data on target species being unavailable), Newell et al. (1987) concluded that a daily intake of 0.11 mg/kg/day was an acceptable intake level for birds. The estimated daily intake (0.057 mg/kg/day) for eagles or other similar piscivorous birds at Crab Orchard Lake is roughly one half this value, and is well below the NOEL of 1.1 mg/kg/day for reproductive toxicity in domestic hens. Thus, for effects to be seen in eagles at this level, they would have to be significantly more sensitive to PCBs than domestic hens and the birds would have to obtain at least 80% of their fish diet from Crab Orchard Lake during both the summer and winter months.

Benthic and bottom feeding organisms may be at risk from ingestion of bottom sediments bearing PCBs, and possibly from

absorption of localized residues of these compounds in water near and beneath the sediment surface. However, these risks cannot be quantified due to lack of data on ingestion rates and bioavailability of these residues, and uncertainty regarding the ability of these compounds to produce acute and chronic effects at low levels. The risk levels evaluated above for piscivorous mammals and birds suggest that wildlife are not subject to unacceptable exposure levels due to contaminants in Crab Orchard Lake fish.

38.4.3 Sedimentation Analysis

As a followup to the Phase II investigation, split-spoon sediment core samples were collected from five locations (32-98, 103, 104, 105, 107, see Figure 36-3) within the embayment adjacent to the Area 9 Landfill. One core was also collected at a location (7E, 34-22) toward the center of the lake, which is further removed from surface discharges and is close to a narrow opening between the west and east lake portions where the water velocity would be greater. The depth to naturally occurring sublake soils was determined for each sediment core by noting the appearance, soil type, and moisture content at 6 in. to 1 ft. intervals for the continuous split-spoon samples. The depths of sediments were measured as 66, 51, 54, 33, and 54 inches respectively, or a mean of 51.6 inches for the Area 9 embayment. The depth for the sediment core from Location 34-22 at the center of the lake was 18 inches. Since the lake has existed for approximately 45 years, an average sediment deposition rate of 1.2 inch per year or 2.9 cm per year is estimated for the embayment, and 0.4 inch or 1.0 cm per year for the mid-lake areas. According to Fink (1987), these are deposition rates characteristic of productive reservoir systems.

Fisher et al. (1983) studied the release rate to overlying water of four polychlorinated biphenyls (tri- and tetrachloro-) contained in contaminated bottom sediments collected from Waukegan Harbor in Illinois. Having determined the rate of flux and the rate of migration through interstitial water in the sediments by molecular diffusion, these authors concluded that even the most mobile of the PCBs studied had an extremely low diffusivity through the sediments. Using these values for upward migration, it was determined that sediment deposition would overtake diffusion as the determining element for PCB release if other disturbing influences such as wave action, and biological disturbances are ignored. For instance, it was concluded that a sediment deposit rate of 0.004 in. or 0.01 cm per year would remove PCBs in a surface sediment layer from communication with overlying water in approximately three years.

In the absence of other factors, therefore, the estimated sediment deposition rate for the bay would remove sediment containing PCB residues from communication with the free water column within less than a year once remediation of on land contaminants has been completed. As noted by Fisher et al. (1983), sediment disturbances such as caused by wind or current, or by fish and benthic organisms could retard this rate of encapsulation of contaminants. Since this region of the lake is frequented by carp and other bottom-feeding fish, it may be more conservative to assume that the upper 5 cm or so of the sediments, constituting the active sediment zone, is being continuously mixed by the action of fish, waves, and benthic organisms. Thus an influx of 2.9 cm of non-contaminated sediment per year into the bay would reduce existing PCB concentrations by about 60 percent annually, by simple dilution, exclusive of biodegradation or other dissipative mechanisms. Thus, three years after

the transport of PCBs to the bay ceased, residues would be reduced to about 6 percent of previous values, and to 1 percent after 5 years.

38.4.4 Analysis of Uncertainties

As has been discussed elsewhere in this report (see above and Section 6.5), several key areas of uncertainty are contained in this risk assessment. To ensure that the assessment does not understate the magnitude of the upper bound of risk, multiple ranges of assumptions (i.e. dietary habits, residue levels, duration of exposure) spanning a possible least case scenario to a possible worst case scenario were employed. The approach used to derive the worst case risk estimates is not intended to estimate the actual risks which may be associated with this site, which are most likely defined by a probability distribution rather than a discrete value. The probability that even a small subset of the human and wildlife populations might begin to meet the set of worst case assumptions used is small. It is unlikely that the level of risks calculated could be detected above background incidence of disease. The upper bound risk estimates are of importance in this case because of the severity of the potential effects, not because they are likely to occur. In addition, the quantitative assessment model assumes that all PCB aroclors are as carcinogenic as the most toxic Aroclor 1260, even though there is reason to conclude that Aroclor 1254, which was the only Aroclor detected in the residues at this site, may be less potent.

Additional uncertainty is introduced due to the limited size of the analytical sampling program for this RI; this degree of uncertainty must also be considered in the interpretation and conclusions derived from the evaluation of this site. Although limited, the RI analytical data are

generally supported by the more extensive database developed as part of ongoing monitoring programs conducted by the Illinois Department of Public Health (IDPH, 1976-1987) and SIU (Kohler, undated). The results from these monitoring data are presented in the Review of Previous Investigations in Section 2.7.

38.5 Preliminary Remedial Alternatives

Three samples of lake waters remote from the Refuge waterworks intake contained part per trillion levels of PCBs; these results were above the Ambient Water Quality Criteria for human health, but only one sample exceeded the concentration standard for protection of aquatic life. Lake waters near the waterworks intake and after treatment did not contain detectable levels of PCBs (detection limit of 0.005 ug/L). The treated water supplies exceeded the criteria for some trihalomethane compounds, likely due to the drinking water requirements for residual chlorine after treatment. Corrective measures have been instituted in the treatment of these waters to reduce trihalomethane level below the standards. In addition, the water samples from some areas of Crab Orchard Lake contained low levels of arsenic and mercury above the ambient water criteria for human health. However, all detected constituents were below the Ambient Water Criteria for protection of aquatic life with the exception of PCBs in one sample from the embayment near Area 9. Sediment samples from the lake contained phthalates and trace semi-volatile compounds; however, the phthalates were also detected in the laboratory blanks. The sediment samples from the Area 9 embayment also contained microgram (ppb) levels of PCBs.

As evaluated in the previous section, the effects of contaminants in lake sediments appear to have resulted in PCB concentrations in some fish from Crab Orchard Lake above the FDA tolerance level for this regulated substance. Two

carp composites (6.4 and 3 mg/kg PCBs; 3.9 and 4.4 mg/kg when re-analyzed) and one bass composite (1.05 mg/kg mercury) collected from the eastern area of the lake exceeded the FDA criteria of 2.0 mg/kg for PCBs and 1.0 mg/kg for mercury.

These data were evaluated in a detailed risk assessment (Section 38.4), however, with the conclusion that the risks to local fishermen and to wildlife do not warrant direct remediation of lake sediments. Based on the review of previous information and the evaluation of current site conditions as part of this investigation, no remedial measures are recommended for Crab Orchard Lake, except for periodic monitoring of lake waters, as detailed in Attachment 1. The attached Monitoring Plan proposes continuing monitoring of waters at this site and other tributaries and drainage routes to Crab Orchard Lake.

Periodic monitoring of lake fish and sediments is also recommended, whether as part of a continuing program conducted by the State of Illinois, or in conjunction with the proposed plan in Attachment 1. Further monitoring of fish residues with a method permitting a lower analytical detection limit would lead to a more accurate quantitative determination of risks associated with consumption of fish containing fillet PCB residues at the lower end of the range observed in the various investigations, in terms of both species analyzed and area of the lake sampled. In addition, any such monitoring should include analyses for PCDD and PCDF congeners in fillet tissue to more accurately determine overall risk from fish consumption.

38.6 Conclusions and Recommendations

It can be concluded that the waters of Crab Orchard Lake generally meet all regulatory standards and criteria for human health and aquatic life with the exception of low concentrations of PCBs, arsenic and mercury in isolated

locations. Some sediments contained low levels of PCBs, phthalates, and other organics.

Given the circumstances of Crab Orchard Lake, including observed fishing patterns, the remedial measures contemplated for various potential PCB sources around the lake, the low levels and limited areas of PCB contamination in lake sediments, the apparent health of the existing aquatic ecosystem, and the natural self-sealing mechanism provided by the lake sedimentation, no remedial measures are recommended for the lake. For these reasons, the lake will not be carried forward into the Feasibility Study.

As with several other sites not being evaluated in the FS, a proposed followup monitoring program for lake waters has been included as Attachment 1 of this report. It is further recommended that lake fish and sediments continue to be monitored on a periodic (e.g. annual) basis following implementation of source control measures.

SECTION 39 - SITE 35, AREA 9 WATERWAY

39.1 Site Description

Site 35 is a low lying spot in an agricultural field to the east of Area 9. (See Figure 39-1). Vegetation does not grow in the area of depression, indicating the potential presence of contaminants.

39.2 Site Investigations

39.2.1 Phase I Site Investigations:

One composite soil sample was collected during Phase I.

39.2.2 Phase II Site Investigations:

No samples were collected during Phase II.

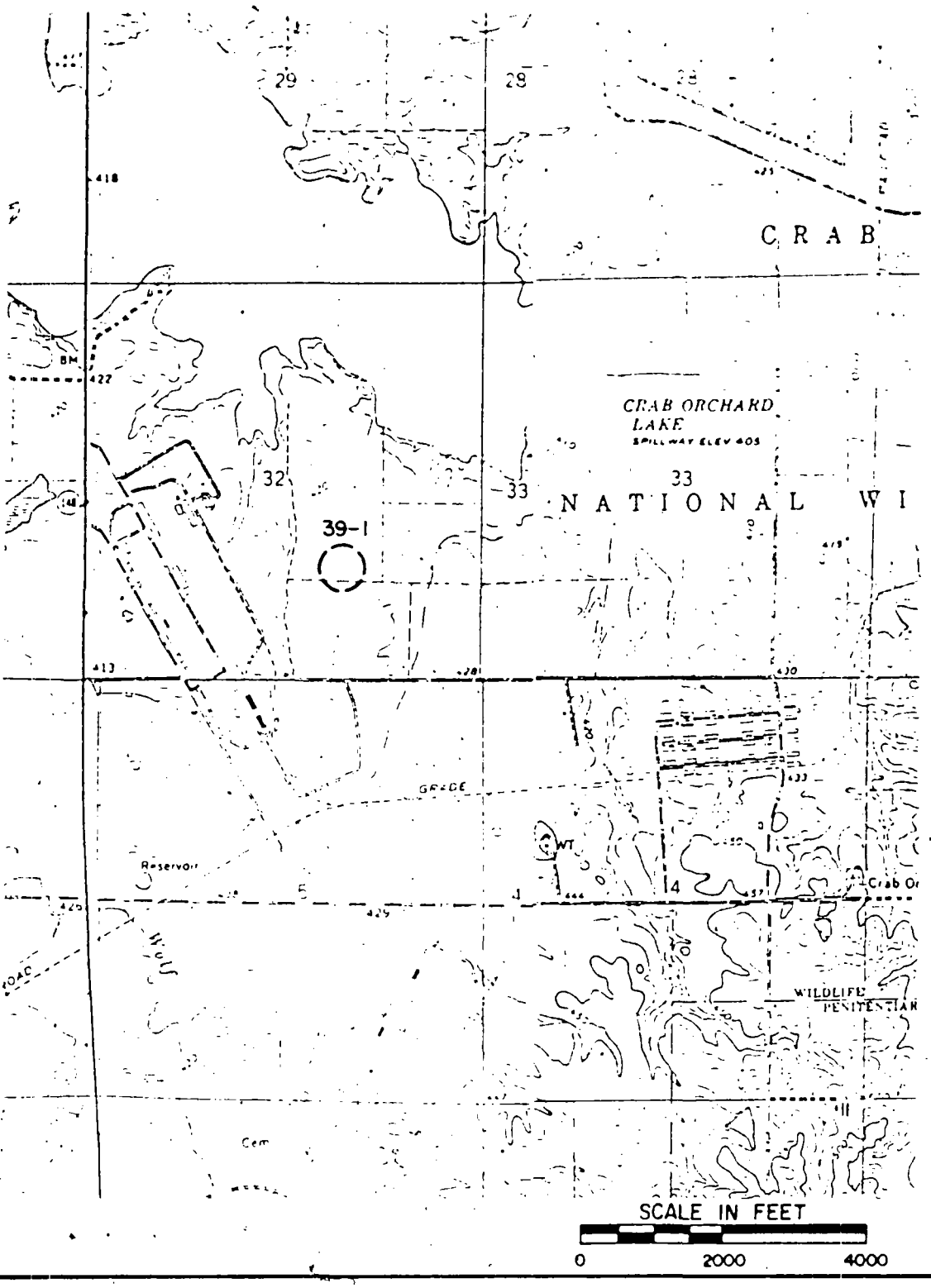
39.3 Analytical Results (See Appendix I, Page 31)

Unusually high specific conductance, 11,650 umhos/cm, was measured in the soil sample, possibly due to the accumulation of salt. A trace (0.016 mg/kg) of PCBs was detected. The concentrations of all other parameters were below those detected at the Refuge control sites, although the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). Some compounds reported as not detected may in fact be present.

FIGURE 39-1

SITE 35
AREA 9 WATERWAY

PHASE I



O'BRIEN & GERE ENGINEERS, INC

F 1 1

39.4 Environmental Effects

39.4.1 Qualitative Assessment

This site was chosen for investigation based on an inspection of the site by the Refuge Manager. The lack of vegetation in the depression of this site led to speculation that this area might be contaminated.

Other than an unusually high specific conductance, which was attributed to the salt content of the soil, all other parameters were within the range of their respective Refuge background levels. Aerial photos show this location to be a wet spot within a low-lying area of an agricultural field. It is speculated that agricultural runoff containing fertilizers, etc. accumulates at this location and then evaporates, leaving behind an accumulation of salts.

Because there is no established waste source at this location, it is not possible to have a "complete" exposure scenario. Therefore, on the basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

39.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

39.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a site inspection and a sample analysis. An inspection of the site revealed a depression in the field that lacked vegetation. It was speculated that contaminants may have caused the lack of vegetation.

Chemical residue information consisted of analytical results for one surface soil sample. This information was obtained only for the top one foot of soil; deeper soil borings were not conducted. Since only a trace of PCBs were detected and all other parameters were below detection limits, there is no evidence to suggest that this site is contaminated.

It can be concluded that the data generated are adequate for evaluation of this site. The sampling analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

39.5 Preliminary Remedial Alternatives

Preliminary Phase I screening results discussed in the previous sections indicated that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore this site was not included in the Phase II investigations. There will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

39.6 Conclusions and Recommendations

It can be concluded that the East Waterway site does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

Respectfully Submitted,

O'BRIEN & GERE ENGINEERS, INC.

C.B. Murphy Jr.

Cornelius B. Murphy, Jr., Ph.D.
Senior Vice President
O'Brien & Gere Project Officer

Prepared by:

Henry T. Appleton, Ph.D., Senior Toxicologist
Donald T. Bussey, Hydrogeologist
Steven R. Garver, P.E., Vice President
David R. Hill, Manager, OBG Laboratories
Swiatoslav W. Kaczmar, Ph.D., Managing Toxicologist
Cynthia M. Klevens, Project Engineer
Mark LaRue, Researcher
Richard G. Stromberg, Sr. Project Hydrogeologist
Matthew Traister, Research Engineer

REFERENCES

Adams, N.R. (FWS). Memorandum to Project Manager, Crab Orchard NWR. Chemical Contamination on Crab Orchard Refuge. (Oct. 13, 1982). 41

Adams, W. Memorandum to Dr. James Edler. Hazardous Waste Sites on Service Lands. (Oct. 17, 1983). 29

Adams, Wayne D. (U.S. DOI). Memorandum to John Ellis, Twin Cities. PCB analyses - Fish Samples in Crab Orchard Lake. (May 30, 1984). 21

Adams, Wayne D. (U.S. DOI). Memorandum to James Gritman, U.S. EPA. Sangamo Electric - Crab Orchard NWR. (May 24, 1984). 23

Adams, Wayne D. (U.S. DOI). Memorandum to John Ellis. Olin's Testing for PCBs in the Sangamo Plant Area. (Jul. 17, 1984). 12

Adams, Wayne D. (U.S. DOI). Memorandum to D. Stalling, CNFRL. Soil and Water Samples - Sangamo - Marion. (Aug. 21, 1984). 8

Agency of Toxic Substances and Disease Registry (ATSDR), Acting Director, Office of Health Assessment, Memorandum to Ms. L. Fabinski, Public Health Advisor, EPA Region V, June 12, 1986. Review of Raw and Finished Water Data, Crab Orchard National Wildlife Refuge (SI-86-097).

American Association of Petroleum Geologists. Geological Map. 1965.

Arnett, C. Ray (U.S. FWS). Letter to Senator Alan Dixon. Presence of PCBs in Crab Orchard Lake Ecosystem. (May 22, 1984). 50

ATSDR (1987). Draft Toxicological Profile for N-nitrosodiphenylamine. October 1987.

ATSDR (1987). Toxicological Profile for Selected PCBs. Agency for Toxic Substances and Disease Registry. Draft. November 1987.

Bacci, E. and Gaggi, C. (1985). Polychlorinated Biphenyls in Plant Foliage: Translocation or Volatilization from Contaminated Soils? Bull. Environ. Contam. Toxico. 35:673-681.

Bandiera, S., Sawyer, T., Romkes, M., Zmudzka, B., Safe, L., Mason, G., Keys, B., and Safe, S. Polychlorinated dibenzofurans (PCDFs): effects of structure on binding to the 2,3,7,8-TCDD cytosolic receptor protein, AHH induction and toxicity. Toxicol. (1984).32:131-144

Bell, Charles R. (Illinois EPA). Memorandum to Ira Markwood. Crab Orchard Lake Sampling. (May 25, 1984). 22

Bell, Charles R. (USEPA, FOS-DPWS, Springfield). Memorandum to Joseph E. Stuart, U.S. EPA, FOS-DPWS, Marion. Williamson County - Crab Orchard Refuge PWS; Facility No. 1997037. (Aug. 8, 1984).

Beyer, W.N., Chaney, R.L., and B.N. Mulhern. 1982. Heavy Metal Concentrations in Earthworms from Soil Amended with Sewage Sludge. J. Environ. Quality. 11:381-385.

Boice, Richard (USEPA Region V). Letter to R. Ruelle, U.S. FWS. Phase I Data QA/QC comments, Crab Orchard RI. (Feb. 18, 1987).

Bradlaw, J.A., Garthoff, L.H., and Hurley, N.E. Comparative induction of aryl hydrocarbon hydroxylase activity in vitro by analogues of dibenzo-p-dioxin. Fd. Cosmet. Toxicol. (1980). 18:627-635.

Bruckner, J.V., Khanna, K.L and Cornish, H.H (1974). Effect of Prolonged Ingestion of Polychlorinated Biphenyls on the Rat. Fd. Cosmet. Toxicol. V12:323-330.

Bush, B., Shane, L.A., Wilson, L.R. Barnard, E.L. and Barnes, D. (1986). Uptake of Polychlorinated Biphenyl Congeners by Purple Loosestrife (Lythrum salicaria) on the Banks of the Hudson River. Arch. Environ. Contam. Toxicol. 15:285-290.

Byram, Scott. Memo to File. Crab Orchard National Wildlife Refuge - Sangamo Dump. TDD No. R5-8308-6. (Sep. 21, 1983). 31

Byram, S. Ecology & Environmental. Potential Hazardous Waste Site - Site Inspection Report. (Oct. 3, 1983). 48

Callahan, M.A. et al. Water Related Environmental Fate of 129 Priority Pollutants, VI. EPA-440/4-79-0292. (Dec., 1979).

Carlson, Richard J. (IEPA). Letter to Wayne Adams, FWS. Notice of Sealing of Site. (May 16, 1984). 24

CDC/ATSDR (1987) Toxicological Profile for Selected PCBs. Agency for Toxic Substances and Disease Registry. Draft. November 1987.

Chew, R.M. (1965). Water Metabolism of Mammals. In Mayer, W.V. and R.G. VanGelder. Physiological Mammalogy, V.2. pp. 43-178. Academic Press. New York.

Chu, C.K., Stella V.J. Bruckner, J.V. and Jiang, W.D. (1977). Effects of Long-Term Exposure to Environmental Levels of Polychlorinated Biphenyls on Pharmacokinetics of Pentobarbital in Rats. V.66, N.2:238-241 (February, 1977).

Clement Associates, Inc. (1985). Chemical, Physical, and Biological Properties of Compounds present at Hazardous Waste Sites, Final Report to U.S. EPA, Arlington, Va. (Sep. 27, 1985).

Contract Laboratory Program for Organic Analysis - Multimedia, Multicomponent, see USEPA citations.

Cordle, F., R, Locke, and J. Springer. 1982. Risk Assessment in a Federal Regulatory Agency: An Assessment of Risk Associated with the Human Consumption of some Species of Fish Contaminated with Polychlorinated Biphenyls (PCBs). Environ. Health Perspec. 4:171-182.

Daniels, W. and Kramkowski, R. (1986). Health Hazard Evaluation Report, National Institute for Occupational Safety and Health. HETA 85-334-1676, Olin Corp., Marion, Illinois. (Mar., 1986).

Davis, S.N. and Dewiest, R.J. (1966). Hydrogeology. John Wiley & Sons.

Dawson, J.P. (Olin Corporation). Memorandum to L. A. Krause. Chlorinated Biphenyls - Marion, Wipe Samples. (Aug. 7, 1984).

Deifenbach, Russ (Dept. of Health and Human Services). Memorandum to Louise A. Fabinski, EPA Region V. Occupational Health Hazard - Olin Corporation, Crab Orchard NWR, Marion, Illinois. (Apr. 14, 1985).

Envirodyne Engineers, Inc. PCB Profile of "I" Area. Prepared for Olin Corporation. (Aug. 1984).

Envirodyne Engineers, Inc. PCB Profile of Three Locations in Area "I". Prepared for Olin Corporation. (Aug. 1984).

Fink L. (1987) Telephone conversation between Larry Fink, Chemist, USEPA Region V, Grosse Isle, MI, and Henry Appleton, Senior Scientist, O'Brien & Gere, Syracuse NY. Nov. 3, 1987.

Fink L. E. (1986). Risk Assessment Calculations for PCBs in Crow Island Dredged Material. Great Lakes National Program Office, Grosse Isle MI. Oct. 1986.

Fisher J.B., R.L. Petty and W. Lick. (1983). Release of Polychlorinated Biphenyls from Contaminated Lake Sediments: Flux and Apparent Diffusivities of Four Individual PCBs. Environ. Pollution (series B) 5:121-132.

Forbis, Leanne (Analytical Bio-Chemistry Laboratories, Inc.). Letter to Glen Clarida, U.S. EPA-DLPC, Marion, Illinois. PCB Analysis in Catfish. (Jun. 15, 1984). 16

Frankland, Les. Memorandum to Bob Cole. Transmittal of Results from 1981 Deer Sampled at Crab Orchard Lake Wildlife Refuge. 51 samples. (Jul. 9, 1984). 54

Frankland, Les (Illinois DOC). Memorandum to Matt Rice, FWS. Crab Orchard Lake Synopsis. (Aug. 21, 1984). 46

Frye, J.C. Outline of Glacial Geology of Illinois and Wisconsin, Quaternary of the United States, Princeton, NJ. (1965)

Gifford, Michael A. (Ecology & Environmental, Inc.). Memorandum to File. Illinois/TDD R5-8308-06A; Crab Orchard National Wildlife Refuge/Sangamo Dump. (Apr. 5, 1984). 27

Gritman, James C. Letter to Dr. David Kenney. Results of Contaminant Analyses on Soil and Fish Samples from Crab Orchard National Wildlife Refuge, sampled Sept. 1982 (No date). 56

Gritman, James C. (U.S. FWS, Twin Cities, Mn). Letter to Valdas V. Adamkus, USEPA, Chicago, IL Hazardous Waste Site at Crab Orchard NWR. (Jun. 1, 1984). 18

Gross, M.L. (Univ. of Nebraska). Letter to R. Ruelle, FWS. PCDD and PCDF Data. (Aug. 26, 1983). 33

Gross, M. (Univ. of Nebraska). Analysis of Polychlorodibenzo-p-dioxin (PCDD) and Polychlorodibenzofuran (PCDF) in Soil Samples. (Oct. 7, 1983). 30

Guyton, A.C. (1947). Measurement of the Respiratory Volumes of Laboratory Animals. American Journal of Physiology 150:70-77.

Harbison, R.D., R.C. James, and S.M. Roberts. (1987). Biological Data Relevant to the Evaluation of Carcinogenic Risk to Humans. Prepared for the Scientific Advisory Panel, Safe Drinking Water Enforcement Act, State of California. (August, 1987). University of Arkansas School of Medicine, Little Rock, Arkansas. See Exhibit C.

Hawley, J.K. (1985). Assessment of Health Risk from Exposure to Contaminated Soil. Risk Analysis V.5:289-302.

Hileman, B. (1988). The Great Lakes Cleanup Effort. Chemical & Engineering News. Washington D.C. (February 1988).

Hill, DR. (1988). Conversation with H.T. Appleton, O'Brien & Gere, Syracuse, NY, May 31, 1988.

Hite, R. L. and King, M. Biological Investigation of the Crab Orchard Creek Basin, Summer 1975. Illinois Environmental Protection Agency. (Apr. 1977). 44

Hite, Robert L. (EPA-DWPC). Letter to Richard Ruelle, FWS. PCB Monitoring in Crab Orchard Lake. (May 8, 1984). 25

Hite, Robert L. and Martin H. Kelly. Staff Report. Polychlorinated Biphenyl Monitoring, Crab Orchard Lake, 1983. (May 1984). 5

HSDB (1987) Hazardous Substances Data Bank. N-nitrosodimethylamine. National Library of Medicine. Bethesda, MD.

Huckins, Jim. (U.S. DOI-FWS). Letter to Wayne Adams, CONWR. Results of Crab Orchard Sample Analyses from Area 11. (Sep. 18, 1984). 45

Hurley, J. (IEPA, DPWS). Report to J. B. Tolson (U.S. Fed. Penitentiary - Marion). Trihalomethane Analysis Report. (May 8, 1984). 51

Hwang S.T., Falco J.W. and Nauman C.H. (1986). Development of Advisory Levels for Polychlorinated Biphenyls (PCBs) Cleanup.

ICF Incorporated. (1985). Superfund Health Assessment Manual, Draft, Washington, D.C., EPA Contract No. 68-01-6872. (May 22, 1985).

Illinois DOA. Memorandum to Les Frankland, IL DOC. Toxicology Report on Fish. (Apr. 4, 1983). 36

Illinois Department of Public Health Monitoring Data for Crab Orchard Lake (1976-1987). Transmittal from R. Boice to O'Brien & Gere, November 2, 1987.

Illinois Environmental Protection Agency. Staff Report. Biological and Water Quality Survey of Crab Orchard Creek in Vicinity of Marion Wastewater Treatment Plant, Marion, Illinois, 1979 and 1980. (Jul. 1981). 42

Kelly, M.H. and Hite, R.L. Chemical Analysis of Surficial Sediments from 63 Illinois Lakes, Summer 1979. Illinois Environmental Protection Agency. (1981). 43

Kelly, M.H. and Hite, R.L. Evaluation of Illinois Stream Sediment Data: 1974-1980. Illinois Environmental Protection Agency. (Jan. 1984).

Kenney, David (IL DOC). Letter to J. Gritman, FWS. Fish Sample Analysis Report for Crab Orchard Lake. (Apr. 18, 1983). 53

Kimbrough, R.D., Linder, R.E. Gains, T.B 91972). Morphological Changes in Livers of Rats Fed Polychlorinated Biphenyls. Arch. Environ. Health V.25:354-364:

Krause, L.A. (Olin Corporation). Memorandum to A. Heinz. Polychlorinated Biphenyl Study - Marion, Preliminary Report. (Jul. 20, 1984).

Linder, R.E. Gains, T.B., Kimbrough, R.D. (1972). The Effect of Polychlorinated Biphenyls on Rat Reproduction. Fd. Cosmet. Toxicol. V.12:63-77.

Lindsay, W.L. Chemical Equilibrium in Soils. John Wiley & Sons. (1979).

Litterst, C.L., Farber, T.M., Baker, A.M. and Van Loon, E.J. (1972). Effect of Polychlorinated Biphenyls on Hepatic Microsomal Enzymes in the Rat. Toxicology and Applied Pharmacology. 23:112-122.

Newell A.J., D.W. Johnson, and L.K. Allen. (1987). Niagara River Biota Contamination Project: Fish Flesh Criteria for Piscivorous Wildlife. NYS Department of Environmental Conservation Publication. (July 1987). Albany NY.

Norback D.H. and R.H. Weltman (1985). Polychlorinated Biphenyl Induction of Hepato-Cellular Carcinoma in the Sprague-Dawley Rat. Environ. Health Perspect. 60:97-105.

NYS Department of Environmental Conservation. Superfund and Contract Laboratory Protocol. (Jun. 1986).

O'Brien & Gere Engineers, Inc. Letter to John N. Hanson, Esq., Beveridge & Diamond. Diefenbach's Memo on Occupational Exposure to PCBs. (Apr. 22, 1985).

O'Brien & Gere Engineers, Inc. Scope of Work. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Jun. 1985).

O'Brien & Gere Engineers, Inc. Work Plan. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Jun. 1985).

O'Brien & Gere Engineers, Inc. Work Plan Supplement. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Dec. 1985).

O'Brien & Gere Engineers, Inc. Work Plan Supplement. Phase II. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Nov. 1986).

O'Brien & Gere Engineers, Inc. Quality Assurance Project Plan. *Revision 4. Remedial Investigation/Feasibility Study.* Crab Orchard National Wildlife Refuge. (Nov. 1986).

O'Brien & Gere Engineers, Inc. Draft Site Investigation Report. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Feb. 1987).

O'Connor, Harold J. (USFWS, Wash. DC). Memorandum to D. Rosenberger, U.S. FWS/RCA. Lead Contamination of White-Tailed Deer at the Crab Orchard National Wildlife Refuge, Illinois. (Jun. 22, 1984). 15

O'Toole, Michael (USEPA). Memorandum to Russell Diefenbach. Crab Orchard Creek National Wildlife Refuge, Marion, Illinois Trip Report. (Jul. 21, 1983). 52

O'Toole, Michael (USEPA, Chicago). Letter to D. Stalling, CNFRL. Transmittal of Fish & Sediment Analysis. (Jul. 17, 1984). 11

O'Toole, Michael (USEPA, Chicago). Memorandum to File. Sampling Effort at the Sangamo Dump on Aug. 15, 1984. (Sep. 17, 1984). 9

Paladino, Pete (IL DOC). Memorandum to Mike Carter. Contaminant Sampling of Crab Orchard Lake.

Palmer, E.L and Fowler, H.S. (1975). Fieldbook of Natural History, Second Ed., McGraw Hill Co., New York.

Paustenbach, D.J., Shu, H.P., and Murray, F.J. Critical Assumptions in Risk Assessment of Soil Contaminated with 2,3,7,8-TCDD (Dioxin). From Workshop on Assessing Risks from Chemical Contaminants in Soil. Andover, MA (May 8, 1986).

Porter, J. Winston (USEPA). Memorandum to EPA Regional Administrators, Regions I-X. Interim RCRA/CERCLA Guidance on Non-Contiguous Sites and On-Site Management of Waste and Treatment Residue. (Mar. 27, 1986).

Redmon, Walter L. (U.S. EPA Reg. V) Memorandum to Kenneth Ferner. Trip Report - Meeting at Crab Orchard National Wildlife Refuge on Contamination of the Refuge with PCBs, Lead and Other Chemicals. (Aug. 11, 1983). 47

Registry of Toxic Effects of Chemical Substances (RTEC). NIOSH. (June, 1983).

Ruelle, D. (1987). Telephone conversation between D. Ruelle, U.S. FWS Project Director, Rock Island, IL and H. Appleton, Senior Scientist, O'Brien & Gere, Syracuse, NY. December 10, 1987.

Ruelle, Richard (USFWS). Letter to C.B. Murphy, OBG Engineers. Comments to Draft RI Report. (Jun. 9, 1987).

Ruelle, Richard (FWS). Report to Refuge Manager. Mercury Levels in Crab Orchard Lake Largemouth Bass. (Feb. 1983). 38

Ruelle, Richard (FWS). Report to Refuge Manager. Survey for Lead on Crab Orchard National Wildlife Refuge. (Feb. 1983). 39

Ruelle, R. and Adams, R. (FWS) Survey for Polychlorinated Biphenyls In Some Abandoned Industrial Dumps and in Lake Sediments on Crab Orchard National Wildlife Regue. (Apr. 1984). 28

Ruelle, Richard (FWS). Memorandum to List. RI/FS Crab Orchard NWR. (Jul. 11, 1984). 13

Ruelle, Richard and Ross Adams (FWS). Survey for Metals in Deer Livers, and in Soils and Vegetation in Abandoned Industrial Dumps on Crab Orchard National Wildlife Refuge. (Jul. 1984).

Safe, S., Bandiera, S., Sawyer, T., Zmudzka, B., Mason, G., Romkes, M., Denomme, A., Sparling, J., Okey, A.B., Fujita, T. Effects of Structure on Binding to the 2,3,7,8-TCDD Receptor Protein and AHH Induction - Halogenated Biphenyls. Environ. Health Perspec. (1985). 61:21-34.

Sawyer, T. and Safe, S. PCB Isomers and Congeners: Induction of Aryl Hydrocarbon Hydroxylase and Ethoxyresorufin o-Deethylase Enzyme Activities in Rat Hepatoma Cells. Toxicol. (1982). Lett 13:87-94.

Sawyer, T.W., Safe, S. In Vitro AHH Induction by Polychlorinated Biphenyl and Dibenzofuran Mixtures: Additive Effects. Chemosphere. (1985). 14:79-84.

Sawyer T.W., Vatcher, A.D., Safe, S. Comparative Aryl Hydrocarbon Hydroxylase Induction Activities of Commercial PCBs in Wistar Rats and Rat Hepatoma H-4-II E cells in culture. Chemosphere. (1984). 13:695-701.

Schmidt, C.J. (1985). National Pesticide Monitoring Program. Arch. Environ. Contam. Toxicol., 14:225-60. As reported in ATSDR (1987).

Shawney, B.L and Hankin, L. (1984). Plant Contamination by PCBs from Amended Soils. J. Food Protection, V.47, N.3:232-236 (March, 1984).

Skea J.C., H.A. Simonin, E.J. Harris et al. 1979. Reducing Levels of Mirex, Aroclor 1254, and DDE by Trimming and Cooking Lake Ontario Brown Trout and Small Mouth Bass, Great Lakes Res. 5:153-159.

Stalling, D. L. Memorandum to ECL Specialist Region III. Dibenzofuran and Dioxin Residues in PCB contaminated Crab Orchard soil. (Sep. 9, 1983). 32

Stalling, David L. Memorandum to Dick Ruelle, Reg. III, RCA Specialist. Quality Control/Quality Assurance Statement for Crab Orchard RFP. (Sep. 24, 1984). 7

Suzuki, M., Aizawa, N., Okano, G. and Takahashi, T. (1977). Translocation of Polychlorinated Biphenyls in Soil into Plants: A Study by a Method of Culture of Soybean Sprouts. Arch. Environ. Contam. Toxicol. 5:343-352.

Thomas, Bob (DWPC). Memorandum to Jim Frank, DLPC. PCBs in Crab Orchard Lake/Sangamo Electric Company Inactive Dump Site. (May 4, 1984). 26

Updike, Gerald H. (FWS). Memorandum to John Ellis, Crab Orchard NWR. Hazardous Waste Site Cleanup Meeting Notes. (Jul. 2, 1984). 49

U.S.D.A. (United States Department of Agriculture). Big Muddy River Comprehensive Basin Study, Appendix K, Agriculture. (1968).

U.S.D.A. Soil Conservation Service. Williamson County Soils, Urbana, Illinois. (1959).

USEPA (1980). Ambient Water Quality Criteria for Polychlorinated Biphenyls. Washington, DC. (Oct., 1980). 881-117798.

USEPA (1984). Remedial Response Program. Hazardous Waste Site Listed Under CERCLA. (Jun. 15, 1984). 17

USEPA. (1984). Health Effects Assessment for Lead. ECAO-CIN-H055. Cincinatti, OH. (Sep., 1984).

USEPA (1985). Contract Laboratory Program Caucus Protocol - Multimedia, Multicomponent. (Jan., 1985).

USEPA (1985). Guidance on Remedial Investigations under CERCLA. Prepared for the Hazardous Waste Engineering Research Laboratory (Office of Research and Development) and Offices of Emergency and Remedial Response, and Waste Programs Enforcement (Office of Solid Waste and Emergency Response). (May, 1985).

USEPA (1985). Reference Values for Risk Assessment. First Draft. ECAO-CIN-477. Environmental Criteria and Assessment, Cincinnati, OH.

USEPA (1986). Superfund Exposure Assessment Manual, Draft. Office of Emergency and Remedial Response, U.S. Environmental Protection Agency. Washington DC. 20460. January, 1986.

USEPA/600/6-86/002 (May 1986). Office of Research and Development. U. S. Environmental Protection Agency. Washington DC 20460.

USEPA (1986). Development of Advisory Levels for PCBs Cleanup. (May, 1986)

USEPA (1986). Health and Environmental Effects Profile for Phthalic Esthers. ECAO-CIN-P188. (Sep., 1986).

USEPA (1986--) Health and Environmental Effects Profile on Nitrosamines. March 1986. ECAO. Cincinnati OH.

USEPA (1987). Memorandum from L.M. Thomas to Administrators and General Counsel. Interim Policy for Assessing Risks of Dioxins Other Than 2,3,7,8-TCDD. (Jan. 7, 1987).

USEPA (1987). Drinking Water Criteria Document for Polychlorinated Biphenyls (PCBs). Prepared by the Environmental Criteria and Assessment Office, U.S. EPA, Cincinnati, OH. May 1987. ECAO-CIN-414

USEPA (1987). OHEA Documentation of ADIs, qs, and Associated Relevant Data. Environmental Criteria and Assessment Office, Cincinnati, OH.

USEPA (1988). Risk Assessment for Dioxin Contamination, Midland, Michigan, Region V. Chicago, IL. EPA-905/4-88-005.

USEPA/Clement Associates, Inc. Chemical, Physical, and Biological Properties of Compounds present at Hazardous Waste Sites. See Clement Associates, Inc.

USEPA-DLPC, Illinois Memorandum. Proposed sampling of public water supplies. (May 31, 1984). 19

USEPA/ICF Inc.. Superfund Health Assessment Manual, Contract No. 68-01-6872. See ICF Incorporated.

Vleck, D. (1979). The Energy Cost of Burrowing by the Pocket Gopher (Thomomys bottae). Physiological Zoology V.52:122-136.

Williams, G.M., and Weisburger, J.H. Chemical Carcinogens. Casarett and Doull's Toxicology. 3rd ed. C.D. Klassen, M.O. Amden and J. Doull, eds. MacMillan Publishing Co. New York, N.Y. (1986).

Wolf, Greg (USFWS). Memorandum to Files. Historical Summary of Contaminants on CONWR. (May 31, 1984). 20

Woolf, Alan et al. Regional Variation in Metals in Livers of White-Tailed Deer in Illinois. Trans. Ill. State Academy of Science. (Jan., 1983). 1, 2:305-310. 40

Zabik M.E., P. Hoojjat, and C.M. Weaver, 1979. Polychlorinated Biphenyls, Dieldrin, and DDT in Lake Trout, Cooked by Broiling, Roasting, or Microwave. Bull, Environ. Contam. & Toxicol. 21:136-143.

Zabik, M.E., C. Merrill, and M.J. Zabik. 1982. PCBs and other Xenobiotics in Raw and Cooked Carp. Bull, Environ. Contam., Toxicol. 28:710-715.

REMEDIAL INVESTIGATION REPORT

CRAB ORCHARD NATIONAL WILDLIFE REFUGE

ATTACHMENT 1

MONITORING PLAN

TABLE OF CONTENTS

TABLE OF CONTENTS
(Continued)

SECTION 1 - INTRODUCTION

1.01 Purpose

This Monitoring Plan has been developed to provide the specifications for periodic routine monitoring surface waters at the Crab Orchard National Wildlife Refuge (CONWR). The purpose of this monitoring program is to ensure that water quality at CONWR does not deviate significantly from the conditions observed during the Remedial Investigation.

Quarterly reports presenting the results of the site monitoring will be prepared and filed at the Refuge headquarters. The quarterly reports are described in Section 3.05. Monitoring activities should continue for a period of thirty years or through the duration of active industrial operations at the Refuge.

1.02 General

The Crab Orchard National Wildlife Refuge is owned by the U.S. government and is currently administered by the U.S. Fish and Wildlife Service (FWS). During the past 40-50 years, several industrial tenants have leased land on the eastern portion of the refuge for manufacturing operations. Some industrial tenants have continued their operations on the eastern area, while the western portion of the refuge is a popular recreation area. Public access to the refuge is limited to authorized personnel on the eastern portion. A remedial investigation (RI) of 33 study sites at the refuge was completed in 1988 to evaluate the the existing conditions of Refuge sites, tributaries of Crab Orchard Lake and Crab Orchard Lake, a drinking water supply for the refuge. Six

sites were carried forward to a Feasibility Study evaluation of remedial alternatives due to the presence of some contaminants in soil or sediments. Eight sites (Table 1) were recommended for monitoring of surface waters due to their proximity to active industrial operations. Section 4 details the site specific monitoring locations and parameters for these eight monitoring sites.

SECTION 2 - SAMPLING PLAN

2.01 Sample Types

Surface water is the media of interest for the purposes of this Monitoring Plan. Waters from surface streams and Crab Orchard Lake will be sampled.

For the most part, all samples will be obtained as single grab samples. However, at some sites, composite samples will be prepared. The compositing procedure to be followed is discussed under the sampling procedures described below for each type of surface water that is to be sampled.

Field blanks, field duplicates and matrix spikes/matrix spike duplicates (MS/MSD) will also be collected or prepared as part of the quality assurance and quality control (QA/QC) requirements outlined in Section 3.

The number of samples to be collected for each of these categories will be dependent on the total number of samples to be collected. A description of each type of QA/QC sample follows. Field duplicates are two distinct samples taken from the same location at similar times using identical sampling equipment that has been decontaminated in a similar manner. One field duplicate will be collected for every ten samples collected. Field blanks for surface waters will be prepared using ultrapure distilled/dionized water. The field blank sample will be placed into the appropriate decontaminated sampling equipment, removed from the equipment and then placed into the proper sampling containers. One field blank will be collected for every twenty samples collected. Matrix spike (MS) samples are collected following the sampling

procedure for the matrix being investigated using the same procedure as for the field duplicate samples. Samples tagged as spikes will be treated with matrix spiking solutions in the laboratory and will be analyzed in duplicate (MS/MSD). One matrix spike sample will be collected for every twenty samples.

2.02 Sampling Procedures

2.02.01 Surface Water

Surface water samples should be taken from 2 to 5 (or more) points spaced equally across the width of the stream, seepage, or pond. The specific number of points may be determined in the field and should be adequate to accurately reflect the size of the body of water being sampled. At each point, subsamples should be collected, representative of the total depth of the body of water. The subsamples may then be composited into a single sample for analysis, dependent upon the intent of the sampling program. For small shallow streams, a single sample collected just below the surface at the stream's midpoint may be adequate for sampling and analyses purposes.

Whether samples are obtained from a boat, a bridge, or by wading into the water body, samples should be taken while facing upstream, away from the influent of the sampler or stream flow.

Collection is accomplished by submerging a precleaned, prelabeled container at the sampling point to the depth required. Sampling containers should be preserved with the appropriate preservatives as detailed in Table 2. Container types are also specified in Table 2 for the different analyses required. For deep

streams, or deep ponds, a Kemmerer, Van Dorn or other sampler specifically designed for this purpose may be used. For shallow (i.e. less than three feet deep) locations, an inverted sample container may be carefully submerged by hand and then slowly allowed to fill.

Samples should be stored in an insulated ice cooler at 4 degrees Celsius. All pertinent information should be recorded in the sampling log book and chain of custody forms, including sample collection date, location and identification.

2.02.02 Lake Water Columns

A composite column water sample from Crab Orchard Lake will be obtained as follows: discrete samples from the surface, mid-depth and approximately six inches from the bottom will be taken using a stainless steel Kemmerer, Van Dorn or other sampler specifically designed for this purpose. The sampling device will be decontaminated and rinsed with the water to be sampled prior to each sample collection.

Equal aliquots from each of the three depths at each site should be composited in the proper precleaned, prelabeled, preserved containers as necessary for the analyses to be completed. The samples should be stored in an insulated ice cooler at 4 degrees Celsius. All pertinent information should be recorded, including sample collection date, location and identification in the sampling log book and chain of custody forms.

2.03 Sample Control

2.03.01 Sample Containers and Preservation

Sample containers, sample preservation and filling instructions may be different for each type of analysis that is to be performed. Care must be taken to utilize the correct sample container(s) and preservative(s) to ensure the integrity of the samples. Table 2 provides a listing of the sample preservatives and the sample containers to be utilized. Sample containers will be supplied by the contract laboratory. The collected samples will be kept out of direct sunlight and, after decontamination and labeling, will be placed in coolers for shipment to the contract laboratory.

2.03.02 Sample Shipment and Chain of Custody

Samples will be packed and labelled according to DOT regulations and protocols. Samples will be shipped via a 24 hour delivery service to the contract laboratory so that the samples can be extracted within allowable time limits.

Chain of custody procedures must be followed closely to ensure that an accurate record of the collection, transport, analysis and disposal of the sample(s) is documented. The chain of custody procedures include field custody, laboratory custody and evidence files and conform with the procedures outlined in NEIC Policies and Procedures (EPA-3309-78-001-R).

Samples are accompanied by a field chain of custody record (Figure 1). When transferring the possession of samples, the individuals relinquishing and receiving will sign, date and note the time on the record. This procedure documents sample custody transfer. Samples will be packaged with a separate custody record

accompanying each shipment. All shipments will be accompanied by the field chain of custody record identifying its contents. The original record will accompany the shipment and a copy of the original record will be retained by the Project Coordinator.

Whenever samples are split with a source or government agency, it is noted in the "Remarks" section of the custody record. The note indicates with whom the samples are being split and is signed by both the sampler and recipient. If either party refuses a split sample, this will be noted and signed by both parties. The person relinquishing the samples to the facility or agency should request the signature of a representative of the appropriate party, acknowledging receipt of the samples. If a representative is unavailable or refuses to sign, this is noted in the "Remarks" space. When appropriate, as in the case where the representative is unavailable, the custody record should contain a statement that the samples were delivered to the designated location at the designated time.

If the samples are sent by mail, the package will be registered with return receipt requested. If sent by common carrier, a Government Bill of Lading will be used. Air freight shipments are sent collect. Freight bills, Post Office receipts and Bill of Lading will be retained as part of the permanent documentation.

All documents and raw data from the individual laboratories performing specific analysis will be transferred at the end of the monitoring period to the Refuge Manager for the Fish and Wildlife Service, CONWR for safekeeping for a period of ten years.

SECTION 3 - ANALYTICAL PROTOCOLS, QUALITY ASSURANCE AND CONTROLS

A brief discussion of analytical protocols, quality assurance and control procedures follows.

3.01 Calibration Procedures

3.01.01 Equipment

Generally, all field equipment will be calibrated in accordance with the manufacturer's instructions. Any field equipment that is not covered by the investigator's standard operating procedure will have a specific calibration and operation instruction sheet prepared for it.

3.01.02 Standards

Standards may be generally grouped into two classifications: primary and secondary. Primary standards include USP drugs, NBS and ASTM materials, and certain designated EPA reference material. All other standards are to be considered secondary. No testing of primary standards is necessary. Secondary standards will be examined when first received, and less stable standards will be rechecked at appropriate intervals, usually six months to one year.

3.01.03 Records

A records book will be kept for each standard. Each record will include name and date received, source, code or lot number,

purity, testing data, special storage requirements and storage location. Records will be kept on each instrument requiring calibration, to record all activities associated with maintenance, QA monitoring and repairs program.

3.02 Analytical Procedures

The analyses and methods detection limits for analytical parameters are given in Table 3. When analyzing samples by the listed standardized methods, the accuracy or precision of the data generated by the laboratory is determined through analyses of replicates, spiked samples, synthetic reference standard samples, and field and laboratory blanks along with each set of samples. Any interference is identified and documented. The required QA/QC samples to be collected are specified in Section 2.

3.03 Internal Quality Control

3.03.01 Analytical Procedures and Laboratory Quality Control

The quality control objectives for the monitoring analytical program are listed in Table 4. The frequency of replicate samples, spiked samples, reference samples and blanks, as well as control limits for acceptability are identified in this table.

Quality control data, which includes the analysis of EPA standard reference materials to verify initial calibration of non-CLP analysis, and reports of blanks, duplicates and spiked samples will be included with each package. The laboratory selected to perform the analytical procedures for this monitoring program will be certified by the appropriate State and Federal Agencies. The

laboratory should participate and meet acceptance criteria established by the Illinois inter-laboratory QA/QC programs for analyses of split samples and spiked standards.

3.03.02 Field Control and Preventive Maintenance

Field sampling crews will be under the direct supervision of a crew chief. Records will be used to document the collection of each sample.

Preventive maintenance procedures will be carried out on all field equipment in accordance with manufacturer's equipment manuals. Any field equipment that is not covered by the standard operating procedures will have a specific maintenance instruction sheet prepared for it.

3.04 Data Assessment and Validation

Data assessment will be based upon instrument tuning criteria, duplicate samples, surrogate recoveries, matrix spikes and matrix spike duplicates. Any data that should be rated as "unacceptable" or "preliminary" will be identified. Corrective actions will be identified if required.

Corrective action procedures are developed on a case- by-case basis. These actions may include:

- Reanalyzing samples if holding time requirements have not been exceeded.
- Altering field or handling procedures.
- Resampling.
- Using a different batch of sample containers.

- Recommending an audit of laboratory procedures.
- Accepting data with acknowledged level of uncertainty.
- Discard data.

3.05 Data Reporting

Each quarterly analytical data submission will contain QA/QC sections that summarize data quality information. The reports will include:

1. Discussion of accuracy, precision, completeness of data and results of performance and system audit specified in Table 4.
2. Discussion of results of data assessment.
3. Data results.
4. Chain of custody forms.

SECTION 4 - SITE SPECIFIC MONITORING PROGRAMS

4.01 Site 10 and Site 11 : Waterworks Drainage Channels

4.01.01 Site Descriptions

Sites 10 and 11 receive various drainage channels leading from active industrial operations within the Olin D and P Areas prior to their discharge to Crab Orchard Lake. These channels discharge near the Refuge Waterworks.

4.01.02 Baseline Parameters

Figure 2 depicts the surface water monitoring locations for Sites 10 and 11. The parameters selected for surface water monitoring at the sites include cyanide, iron, magnesium, manganese, mercury and phthalate esters. The results of the RI sampling at Sites 10 and 11 for the previously mentioned parameters are contained in Table 5 and will serve as baseline concentrations for the monitoring program at the sites.

4.01.03 Monitoring

Locations 10-1 and 10-3 at Site 10 and locations 11-1 and 11-3 at Site 11 will be the surface water monitoring locations for the two sites (Figure 2). Future monitoring at Sites 10 and 11 requires that composite surface water samples be collected quarterly from each location at each site. In addition, a full volatile and semi-volatile scan will be run annually on composite surface water samples collected from the four monitored locations at Sites 10 and 11. The samples should be collected and analyzed for the

parameters listed in Section 4.01.2 and in accordance with the procedures outlined in Sections 2 and 3.

4.02 Site 14 : Area 14 Solvent Storage Ditch

4.02.01 Site Description

Site 14 is a drainage ditch adjacent to the active manufacturing operations of Diagraph-Bradley. The ditch receives run-off from a manufacturing area where solvents are handled in bulk or in drums. The ditch runs north, parallel to the road that is west of the plant, and ultimately discharges to Crab Orchard Lake.

4.02.02 Baseline Parameters

Figure 3 and 4 depicts the surface water monitoring locations for Site 14. The parameters selected for surface water monitoring at the site include acetone, chloromethane, methylene chloride and phthalate esters. The results of the RI sampling at Site 14 for the previously mentioned parameters are contained in Table 6 and will serve as baseline concentrations for the monitoring program at the site.

4.02.03 Monitoring

Locations 14-1 and 14-4 at the Solvent Storage Area will be the surface water monitoring locations for Site 14 (Figure 3 and 4). Future monitoring at Site 14 requires that grab surface water samples be collected quarterly from each monitoring location at the site. The samples should be collected and analyzed for the

parameters listed in Section 4.02.02 and in accordance with the procedures outlined in Sections 2 and 3.

4.03 Site 16 : Area 7 Industrial Site

4.03.01 Site Description

Site 16 consists of a ditch within the Area 7 Industrial Site. The Area 7 Industrial Site is comprised of 33 buildings over an area of 55 acres which have been used for a variety of industrial purposes during the last forty years. Three of the buildings are used by Pennzoil in waste oil recovery and recycling operations and two other buildings are used by a refurbisher of mining equipment. The other buildings at the site are used for storage purposes or are abandoned. The Site 16 drainage ditch runs from south to north through the buildings and discharges to Crab Orchard Lake.

4.03.02 Baseline Parameters

Figure 5 depicts the surface water monitoring locations for Site 16. The parameters selected for monitoring at the site include chloromethane, carbon tetrachloride, aldrin, dieldrin and polynuclear aromatic hydrocarbons (PAH) in water. The results of the RI sampling at Site 16 for the previously mentioned parameters are contained in Table 7 and will serve as baseline concentrations for the monitoring program at the site.

4.03.03 Monitoring

Location 16-18 at the Area 7 Industrial Site will be the surface water monitoring location for the site (Figure 5). Future monitoring at Site 16 requires that composite surface water samples be collected quarterly from the single monitoring location at the site. This sample should be collected and analyzed for the parameters listed in Section 4.03.2 and in accordance with the procedures outlined in Sections 2 and 3.

4.04 Sites 25, 26, 27: Crab Orchard Creek

4.04.01 Site Descriptions

Site 25 consists of the Crab Orchard Creek upstream and downstream of the Marion Landfill as well as the adjacent pond. The Old Marion Landfill is located adjacent to Crab Orchard Creek on Old Creal Springs Road. This municipal landfill has been inactive for a number of years. A 3/4 acre pond is located next to the landfill.

Site 26 is located on Crab Orchard Creek downstream from the Marion Sewage Treatment Plant.

Site 27 is located on Crab Orchard Creek downstream of Interstate Route 57.

Sites 25 and 26 are not located within the boundaries of the Refuge and are not under the management of the U.S. FWS. Site 27 is located on the Refuge.

4.04.02 Baseline Parameters

Figure 6 identifies the surface water monitoring locations for Sites 25, 26 and 27. The parameters selected for surface water

monitoring at the sites include cyanide, magnesium, manganese, TOC and TOX in water. The results of the RI sampling at Sites 25, 26 and 27 for the previously mentioned parameters are contained in Table 8 and will serve as the baseline concentrations for the monitoring program at the sites.

4.04.03 Monitoring

Location 25-1, downstream of the municipal landfill location and location 27-1, below the I- 57 dredge area, will be the surface water monitoring locations for the sites (Figure 6). Future monitoring at Sites 25, 26 and 27 requires that composite surface water samples be collected quarterly from each monitoring location at each site. The samples should be collected and analyzed for the parameters listed in Section 4.04.2 and in accordance with the procedures outlined in Sections 2 and 3.

4.05 Site 34: Crab Orchard Lake

4.05.01 Site Description

Site 34 is comprised of Crab Orchard Lake. This lake was formed in 1940 by construction of a spillway across Crab Orchard Creek. The lake has a surface area of 6,965 acres, an average depth of 8-9 feet on the western portion and an average depth of two feet on the eastern portion. The retention time is 0.8 years, with a storage capacity of 72,525 acre-feet.

Water enters the lake through several creeks, including the Crab Orchard Creek at the eastern end of the lake. The eastern section of the lake has been bordered by manufacturing operations

since the 1940s. Water exits the lake through the spillway at the western end and through use of 280,000 gpd of lake water by the Refuge.

4.05.02 Baseline Parameters

Figure 7 identifies the surface water monitoring locations for Site 34. The parameters selected for monitoring at the site include arsenic and PCBs. The results of the RI sampling at Site 34 for the previously mentioned parameters are contained in Table 9 and will serve as the baseline concentrations for the monitoring program at the site.

4.05.03 Monitoring

Locations 34-6, 34-12, 34-14 and 34-15 will be the surface water monitoring locations for the site (Figure 7). Future monitoring at Site 34 require that composite water columns be collected quarterly from each location at the lake at three depths: the surface, mid-depth and approximately six inches from the bottom. The sample should be collected and analyzed for the parameters listed in Section 4.04.2 and in accordance with the procedures detailed in Sections 2 and 3.

TABLES

TABLE 1

MONITORING PLAN
STUDY SITES

SITES 10 and 11 : WATERWORKS DRAINAGE CHANNELS

SITE 14 : AREA 14 SOLVENT STORAGE DITCH

SITE 16 : AREA 7 INDUSTRIAL SITE DITCH

SITES 25, 26 AND 27 : CRAB ORCHARD CREEK

SITE 34 : CRAB ORCHARD LAKE

TABLE 2

SAMPLE PRESERVATION AND CONTAINERS

PARAMETER	WATER
Volatiles	40 ml vial (2)/4 deg.C
Base/Neutrals/Acids	1 liter glass
Pesticide/PCB	1 quart glass (teflon)
PCBs Low Level (water)	1 quart glass (teflon)
Metals	1 pint plastic/HNO3
Cyanide	1 pint plastic/NaOH
Indicators - pH	1 pint plastic

TABLE 3 ATTACHMENT

PROCEDURES FOR LOW-LEVEL PCBs IN WATER

5.0 EXTRACTION PROCEDURE FOR LOW LEVEL PCBs IN WATER

5.1 Determine pH of sample and adjust to a range of 5-9 with .1:1 sulfuric acid solution or in sodium hydroxide. Transfer the entire contents of sample bottle into a 2 liter separatory funnel.

5.2 Add 1ml of 50ppt DBC or equivalent (compound which will not be removed by cleanup options) surrogate solution to the sample.

5.3 To every batch of samples for low level analysis, add a blank, a matrix spike (MS) and a matrix spike duplicate (MSD). The blank is 2000ml of organic-free water and it is treated in the same manner as the samples. The MS and MSD samples are spiked in duplicate. Ideally, a sample is collected in triplicate in the field in 3 separate containers. The first sample is the sample itself. The second and third samples are spiked with the following compound in acetone:

Note: See Tables 4 a 5 for detection level and QA/QC requirement

Aroclor 1254 5ppt

5.4 Add 50ml of 15% methylene chloride in hexane to the sample jar, seal and shake. Transfer bottle extract to separatory funnel and extract the sample by shaking vigorously for 2 minutes with periodic venting to release pressure.

5.5 Drain water sample back into sample jar. Drain the hexane extract through a sodium sulfate funnel into a Kuderna-Danish evaporator. Return water sample back to separatory funnel and repeat extraction 2 more times combining all extracts into the Kuderna-Danish evaporator.

5.6 If emulsion problems occur during extraction, collect all 3 extracts in a 200ml centrifuge bottle without sodium sulfate drying. Centrifuge the contents of the bottle until 2 distinct layers are formed. Transfer the top layer (the hexane extract) through sodium sulfate into the Kuderna-Danish evaporator.

5.7 Concentrate extract to 0.2ml.

5.8 If cleanup is necessary, refer to Section 7.0

5.9 Concentrate final extract after cleanup to 0.2ml and analyze by GC/ECD.

LOW LEVEL PCBs

EXTRACTION PROCEDURE

FOR WATER MATRICES

August 1986

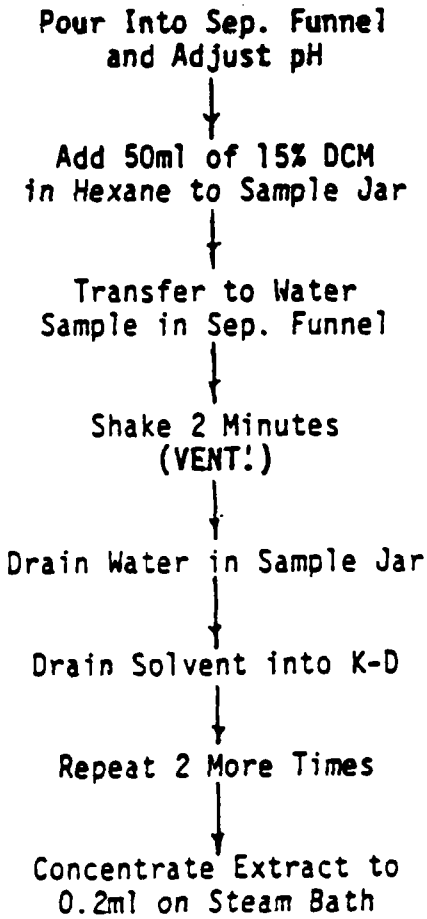


TABLE 4

QUALITY ASSURANCE REQUIREMENTS

ORGANICS

AUDIT	FREQUENCY	CONTROL LIMITS
Reagent Blank	1 per case or 10% of sample shipment.	Less than CRQL.
Field Blank	5% of sample shipment provided by sampling crew.	Less than CRQL.
Matrix Spike	10% of similar concentration/ matrix.	As specified by 40 CFR 136.
Matrix Spike Duplicate	10% of similar concentration/ matrix.	Recoveries as specified in method.

INORGANICS

AUDIT	FREQUENCY	CONTROL LIMITS
Reagent Blank	1 per case or 10% of sample shipment.	Less than CRQL.
Field Blank	5% of sample shipment provided by sampling crew.	Less than CRQL.
Field Duplicate	10% of sample shipment provided by sampling crew.	RPD within 20%.
Matrix Spike	10% of similar concentration/ matrix.	Recoveries as specified in method.

CRQL = Contract Required Quantitation Limit

Table 3

ANALYTICAL METHODS
VOLATILES

PARAMETER	METHOD (1)	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
acetone	602	5	See Table 4
carbon tetrachloride	601	10	"
chloromethane	601	10	"
methylene chloride	601	10	"

(1) Code of Federal Regulations, Title 40, Section 136, Appendix A, "Test Procedure for Analysis of Organic Pollutants."

ANALYTICAL METHODS
SEMI-VOLATILES

PARAMETER	METHOD (1)	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
acenaphthene	610	5	See Table 4
acenaphthylene	610	5	"
anthracene	610	5	"
benzo(a)anthracene	610	5	"
benzo(b)fluoranthene	610	5	"
benzo(k)fluoranthene	610	5	"
benzo(a)pyrene	610	5	"
benzo(g,h,i)perylene	610	5	"
bis (2-ethylhexyl) phthalate	610	5	"
butyl benzyl phthalate	610	5	"
chrysene	610	5	"
dibenzo(a,h)anthracene	610	5	"
diethylphthalate	610	5	"
dimethyl phthalate	610	5	"
di-n-butyl phthalate	610	5	"
di-n-octylphthalate	610	5	"
fluorene	610	5	"
fluoranthene	610	5	"
indeno(1,2,3-cd)pyrene	610	5	"
naphthalene	610	5	"
phenanthrene	610	5	"
pyrene	610	5	"

(1) Code of Federal Regulations, Title 40, Section 136, Appendix A, "Test Procedure for Analysis of Organic Pollutants."

ANALYTICAL METHODS
PESTICIDES/PCBS

PARAMETER	METHOD (1)	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
aldrin	608	50	See Table 4
dieldrin	608	50	"
arochlor-1016	*	0.005	"
arochlor-1221	*	0.005	"
arochlor-1232	*	0.005	"
arochlor-1242	*	0.005	"
arochlor-1248	*	0.005	"
arochlor-1254	*	0.005	"
arochlor-1260	*	0.005	"

(1) Code of Federal Regulations, Title 40, Section 136, Appendix A, "Test Procedure for Analysis of Organic Pollutants."

* See Attachment 1

Table 3
Page 4 of 5

ANALYTICAL METHODS
METALS AND CYANIDE

PARAMETER	METHOD (2)	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
arsenic -T	206.2	10	See Table 4
iron -T	236.1	10	"
magnesium -T	242.1	10	"
manganese -T	243.1	10	"
mercury -T	245.1	0.5	"
cyanide	335.2	50	"

(2) Methods for Chemical Analysis of Water and Wastes, EPA-600/4-79-020

Table 3
Page 5 of 5

ANALYTICAL METHODS
INDICATORS

PARAMETER	METHOD	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
total organic carbon	415.1 (2)	1000	See Table 4
total organic halides	450.1 (3)	10	See Table 4

(2) Methods for Chemical Analysis of Water and Wastes, EPA-600/4-79-020

(3) EPA Method Study 32, EPA/600/54-55/080 (NTIS: PB86136538/AS)

TABLE 5

SITE BASELINE CONCENTRATIONS
SITE 10 AND SITE 11
WATERWORKS DRAINAGE CHANNELS

Baseline Concentrations In Surface Waters

		Site 10		Site 11	
PARAMETER	UNITS	10-1	10-3	11-1	11-3
Cyanide	ug/l	<0.05	<0.05	<0.05	<0.05
Iron (total)	ug/l	600	-	<300	-
Magnesium (total)	mg/l	9.85	-	10.6	-
Manganese (total)	mg/l	0.27	-	0.095	-
Mercury	ug/l	<0.5	-	<0.5	<0.4
Bis (Ethylhexyl) Phthalate	ug/l	-	<20 DR	-	-
Butyl Benzyl Phthalate	ug/l	-	<20 DR	-	-
Di-N-Butyl Phthalate	ug/l	-	<20 DR	-	-
Di-N-Octyl Phthalate	ug/l	-	<20 DR	-	-
Diethyl Phthalate	ug/l	-	<20 DR	-	-
Dimethyl Phthalate	ug/l	-	<20 DR	-	-
Acid Extractable Screen	ug/l	<100	-	<100	-
Base Neutral Screen	ug/l	<100	-	<100	-

KEY:
(-) = Analysis Not Run.
D = Holding time exceeded for extraction.
R = MS/MSD % recovery outside of QC limits.

TABLE 6

SITE BASELINE CONCENTRATIONS
SITE 14
AREA 7 SOLVENT STORAGE DITCH

Baseline Concentrations in Surface Waters

PARAMETER	UNITS	Locations 14-1	14-3	14-5	14-5 (Dup)
Methylene Chloride	ug/l	<1	<1	4 BI	15 BI
Acetone	ug/l	-	-	43 BI	36 BI
Chloromethane .	ug/l	<1	<1	<4 I	<4 I
Bis (Ethylhexyl) Phthalate	ug/l	-	-	<49 R	<20 R
Butyl Benzyl Phthalate	ug/l	-	-	<49 R	<20 R
Di-N-Butyl Phthalate	ug/l	-	-	<49 R	<20 R
Di-N-Octyl Phthalate	ug/l	-	-	<49 R	<20 R
Diethyl Phthalate	ug/l	-	-	<49 R	<20 R
Dimethyl Phthalate	ug/l	-	-	<49 R	<20 R
Acid Extractable Screen	ug/l	<100	<100	-	-
Base Neutral Screen	ug/l	<100	<100	-	-

KEY:
(-) = Analysis Not Run.
B = Indicates that the analyte was found in the blank; the level found in the sample was less than 10 times the level found in the blank or or, additionally in the case of VOAs the level found in the sample was less than 60 ppb of methylene chloride or acetone, which were found in the blank.
I = MS/MSD RPD outside QC limits.
R = MS/MSD % recovery outside of QC limits.

TABLE 8

SITE BASELINE CONCENTRATIONS
SITES 25, 26 and 27
CRAB ORCHARD CREEK

Baseline Concentrations In Surface Waters

		Locations					
PARAMETER	UNITS	25-3 Upstream of Marion Landfill	25-5 Pond Adjacent to Marion Landfill	25-1 Downstream of Marion Landfill	26-3 Upstream of Marion STP	26-1 Downstream of Marion STP	27-1 Below I-57 Dredge Area
Cyanide	mg/l	<0.05 C	-	<0.05 C	<0.05 C	<0.05 C	<0.05 C
Magnesium (total)	mg/l	14.3	5.6	47.2	18.2	26.5	12.4
Manganese (total)	mg/l	0.68	0.72	1.5	0.745	0.30	0.64
TOC	mg/l	11.0	18.0	7.0	0.6	5.0	8.0
TOX	mg/l	0.018	0.011	0.012	0.049	0.120	0.043

KEY:
(-) = Analysis not run.
C = Usable for qualitative interpretation only.

TABLE 7

SITE BASELINE CONCENTRATIONS
SITE 16
AREA 7 INDUSTRIAL SITE

Baseline Concentrations In Surface Waters

PARAMETER	UNITS	Locations 16-1	16-3	16-18
Chloromethane	ug/l	<1	<1	<4 A
Carbon Tetrachloride	ug/l	<1	<1	<4 A
Aldrin	ug/l	<10	<10	0.17 RIN
Dieldrin	ug/l	<10	<10	0.54 RIN
Acenaphthylene	ug/l	-	-	<38 DI
Acenaphthene	ug/l	-	-	<38 DI
Anthracene	ug/l	-	-	<38 DI
Benzo(a)anthracene	ug/l	-	-	<38 DI
Benzo(a)pyrene	ug/l	-	-	<38 DI
Benzo(b)fluoranthene	ug/l	-	-	<38 DI
Benzo(ghi)perylene	ug/l	-	-	<38 DI
Benzo(k)fluoranthene	ug/l	-	-	<38 DI
Chrysene	ug/l	-	-	<38 DI
Dibenzo(a,h)anthracene	ug/l	-	-	<38 DI
Fluoranthene	ug/l	-	-	<38 DI
Fluorene	ug/l	-	-	<38 DI
Indeno(1,2,3-cd)pyrene	ug/l	-	-	<38 DI
Naphthalene	ug/l	-	-	<38 DI
Phenanthrene	ug/l	-	-	<38 DI
Pyrene	ug/l	-	-	<38 DI
Acid Extractable Screen	ug/l	<100	<100	-
Base Neutral Screen	ug/l	<100	<100	-

Key:
(-) = Analysis Not Run.
A = Holding time exceeded for analysis.
D = Holding time exceeded for extraction.
I = MS/MSD RPD outside QC limits.
N = % breakdown level of DDT/Endrin exceeded in previous Eval Mix B-Pest/PCB
R = MS/MSD % recovery outside of QC limits.

TABLE 9

SITE BASELINE CONCENTRATIONS
SITE 34
CRAB ORCHARD LAKE

Baseline Concentrations In Water Columns

Locations

PARAMETER	UNITS	34-6 1(B)	34-7 2(C)	34-8 3(G)	34-9 4(N)	34-10 5(A)	34-11 6(D)	34-12 7(E)	34-13 8(F)	34-14 9(I)	34-15 10(J)
Arsenic (total)	ug/l	3.4 M	2.7	3.2 M	<2.5 M	<2.5 M	<2.5 M	2.00	2.00	<2.5 M	<2.00
PCBs	ug/l	0.008	0.019	<0.005	<0.005	<0.005	0.009	<0.005	<0.005	<0.005	<0.005

KEY:
(-) = Analysis not run.
M = Values determined by a one point standard addition.

LEAD

Summary

Lead is a heavy metal that exists in one of three oxidation states, 0, +2, and +4. There is suggestive evidence that some lead salts are carcinogenic inducing kidney tumors in mice and rats. Lead is also a reproductive hazard, and it can adversely affect the brain and central nervous sytem by causing encephalopathy and peripheral neuropathy. Chronic exposure to low levels of lead can cause subtle learning disabilities in children. Exposure to lead can also cause kidney damage and anemia, and it may have adverse effects on the immune system. The EPA Ambient Water Quality Criterion for the protection of human health and the Interim Primary Drinking Water Standard are both 50 µg/liter.

CAS Number: 7439-92-1

Chemical Formula: Pb

IUPAC Name: Lead

Chemical and Physical Properties

Atomic Weight: 207.19

Boiling Point: 1,740°C

Melting Point: 327.502°C

Specific Gravity: 11.35 at 20°C

Solubility in Water: Insoluble; some organic compounds are soluble

Solubility in Organics: Soluble in HNO_3 and hot concentrated H_2SO_4

Transport and Fate

Some industrially produced lead compounds are readily soluble in water. However, metallic lead and the common lead minerals are insoluble in water. Natural compounds of lead are not usually mobile in normal surface or groundwater because the lead leached from ores becomes adsorbed by ferric hydroxide or combines with carbonate or sulfate ions to form insoluble compounds.

Movement of lead and its inorganic and organolead compounds as particulates in the atmosphere is a major environmental transport process. Lead carried in the atmosphere can be removed by either wet or dry deposition. Although little evidence is available concerning the photolysis of lead compounds in natural waters, photolysis in the atmosphere occurs readily. These atmospheric processes are important in determining the form of lead entering aquatic and terrestrial systems.

The transport of lead in the aquatic environment is influenced by the speciation of the ion. Lead exists mainly as the divalent cation in most unpolluted waters and becomes adsorbed into particulate phases. However, in polluted waters organic complexation is most important. Volatilization of lead compounds probably is not important in most aquatic environments.

Sorption processes appear to exert a dominant effect on the distribution of lead in the environment. Adsorption to inorganic solids, organic materials, and hydrous iron and manganese oxides usually controls the mobility of lead and results in a strong partitioning of lead to the bed sediments in aquatic systems. The sorption mechanism most important in a particular system varies with geological setting, pH, Eh, availability of ligands, dissolved and particulate ion concentrations, salinity, and chemical composition. The equilibrium solubility of lead with carbonate, sulfate, and sulfide is low. Over most of the normal pH range, $PbCO_3$, and $PbSO_4$ control solubility of lead in aerobic conditions, and PbS and Pb control solubility in anaerobic conditions. Lead is strongly complexed to organic materials present in aquatic systems and soil. Lead in soil is not easily taken up by plants, and therefore its availability to terrestrial organisms is somewhat limited.

Bioaccumulation of lead has been demonstrated for a variety of organisms, and bioconcentration factors are within the range of 100-1,000. Microcosm studies indicate that lead is not biomagnified through the food chain. Biomethylation of lead by microorganisms can remobilize lead to the environment. The ultimate sink of lead is probably the deep oceans.

Health Effects

There is evidence that several lead salts are carcinogenic in mice or rats, causing tumors of the kidneys, as a result of either oral or parenteral administration. Data concerning the carcinogenicity of lead in humans are inconclusive. Available data are not sufficient to evaluate the carcinogenicity of organic lead compounds or metallic lead. There is equivocal evidence that exposure to lead causes genotoxicity in humans and animals. Available evidence indicates that lead presents a hazard to reproduction and exerts a toxic effect on conception, pregnancy, and the fetus in humans and experimental animals.

Many lead compounds are sufficiently soluble in body fluids to be toxic. Exposure of humans or experimental animals to lead can result in toxic effects in the brain and central nervous system, the peripheral nervous system, the kidneys, and the hematopoietic system. Chronic exposure to inorganic lead by ingestion or inhalation can cause lead encephalopathy, and severe cases can result in permanent brain damage. Lead poisoning may cause peripheral neuropathy in adults and children, and clinically undetectable permanent-learning disabilities in children may be caused by exposure to relatively low levels. Short-term exposure to lead can cause reversible kidney damage, but prolonged exposure at high concentrations may result in progressive kidney damage and possibly kidney failure. Anemia, due to inhibition of hemoblobin synthesis and a reduction in the life-span of circulating red blood cells, is an early manifestation of lead poisoning. Several studies with experimental animals suggest that lead may interfere with various aspects of the immune response.

Toxicity to Wildlife and Domestic Animals

Freshwater vertebrates and invertebrates are more sensitive to lead in soft water than in hard water. At a hardness of about 50 mg/liter $CaCO_3$, the median effect concentrations for nine families range from 140 µg/liter to 236,600 µg/liter. Chronic values for Daphnia magna and the rainbow trout are 12.26 µg/liter and 83.08 µg/liter, respectively, at a hardness of about 50 mg/liter. Acute-chronic ratios calculated for three freshwater species ranged from 18 to 62. Bioconcentration factors, ranging from 42 for young brook trout to 1,700 for a snail, were reported. Freshwater algae show an inhibition of growth at concentrations above 500 µg/liter.

Acute values for twelve saltwater species range from 476 µg/liter for the common mussel to 27,000 µg/liter for the soft-shell clam. Chronic exposure to lead causes adverse effects in mysid shrimp at 37 µg/liter, but not at 17 µg/liter. The acute-chronic ratio for this species is 118. Reported bioconcentration factors range from 17.5 for the Quahog clam to 2,570 for the blue mussel. Saltwater algae are adversely affected at approximate lead concentrations as low as 15.8 µg/liter.

Although lead is known to occur in the tissue of many free-living wild animals, including birds, mammals, fishes, and invertebrates, reports of poisoning usually involve waterfowl. There is evidence that lead, at concentrations occasionally found near roadsides and smelters, can eliminate or reduce populations of bacteria and fungi on leaf surfaces and in soil. Many of these microorganism play key roles in the decomposer food chain.

Cases of lead poisoning have been reported for a variety of domestic animals, including cattle, horses, dogs, and cats. Several types of anthropogenic sources are cited as the source of lead in these reports. Because of their curiosity and their indiscriminate eating habits, cattle experience the greatest incidence of lead toxicity among domestic animals.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed Criteria)

The concentrations below are for active lead, which is defined as the lead that passes through a 0.45-μm membrane filter after the sample is acidified to pH 4 with nitric acid.

Freshwater

Acute toxicity: $e^{(1.34[\ln(\text{hardness})] - 2.014)}$ μg/liter

Chronic toxicity: $e^{(1.34[\ln(\text{hardness})] - 5.245)}$ μg/liter

Saltwater

Acute toxicity: 220 μg/liter
Chronic toxicity: 8.6 μg/liter

Human Health

Criterion: 50 μg/liter

Primary Drinking Water Standard: 50 μg/liter

NIOSH Recommended Standard: 0.10 mg/m^3 TWA (inorganic lead)

OSHA Standard: 50 $\mu g/m^3$ TWA

ACGIH Threshold Limit Values:

0.15 mg/m^3 TWA (inorganic dusts and fumes)
0.45 mg/m^3 STEL (inorganic dusts and fumes)

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

DOULL, J., KLAASSEN, L.D., and AMDUR, M.O., eds. 1980. Casarett and Doull's Toxicology: The Basic Science of Poisons. 2nd ed. Macmillan Publishing Co., New York. 778 pages

INTERNATIONAL AGENCY FOR RESEARCH ON CANCER (IARC). 1980. IARC Monographs on the Evaluation of Carcinogenic Risk of Chemicals to Humans. Vol. 23: Some Metals and Metallic Compounds. World Health Organization, Lyon, France. Pp. 325-415

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

NRIAGU, J.O., ed. 1978. The Biogeochemistry of Lead in the Environment: Part B. Biological Effects. Elsevier/North-Holland Biomedical Press, New York. 397 pages

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1977. Air Quality Criteria for Lead. Office of Research and Development, Washington, D.C. December 1977. EPA-600/8-77-017

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Lead. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-057

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1983. Draft Revised Section B of Ambient Water Quality Criteria for Lead. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. August 1983

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Lead. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H055 (Final Draft)

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2,332 pages

WORLD HEALTH ORGANIZATION. 1977. Environmental Health Criteria: 3. Lead. World Health Organization, Geneva. 160 pages

Summary

Copper is among the more mobile metals in the environment. It is toxic to humans at high levels; it causes irritation following acute exposure and anemia following chronic exposure. Sheep are very susceptible to copper toxicosis as are many aquatic organisms. The EPA Ambient Water Quality Criterion for copper is 1 mg/liter based solely on its organoleptic propert

Background Information

Copper exists in a valence state of +1 or +2. It is a lustrous, reddish metal. The physical properties of copper include ductility and conductivity of heat and electricity. Copper is found in nature as sulfide, oxide, or carbonate ore.

CAS Number: 7440-50-8

Chemical Formula: Cu

IUPAC Name: Copper

Chemical and Physical Properties

Atomic Weight: 63.546

Boiling Point: 2567°C

Melting Point: 1083°C

Specific Gravity: 8.92

Solubility in Water: Most copper salts are insoluble with the exception of $CuSO_4$, $Cu(NO_3)_2$, and $CuCl_2$ (the more common copper salts). The metal is insoluble in water.

Vapor Pressure: 1 mm Hg at 1628°C

Transport and Fate

Copper has two oxidation states, +1 (cuprous) and +2 (cupri Cuprous copper is unstable in aerated water over the pH range of most natural waters (6 to 8) and oxidizes to the cupric state. Several processes determine the fate of copper in the

humic substances; sorption to hydrous metal oxides, clays, and organic materials; and bioaccumulation. In waters polluted with soluble organic material, complexation with organic ligands can occur, thus favoring the prolonged dispersion of copper in solution. The presence of organic acids also can lead to the mobilization of copper from the sediments to solution. Copper has a strong affinity for hydrous iron and manganese oxides, clays, carbonate minerals, and organic matter. Sorption to these materials, both suspended in the water column and in the sediment, results in relative enrichment of the solid phase and reduction in dissolved levels. Sorption processes are quite efficient in scavenging dissolved copper and in controlling its mobility in natural unpolluted streams. The amounts of the various copper compounds and complexes that actually exist in solution depend on the pH, temperature, alkalinity, and concentrations of other chemical species. The levels of copper able to remain in solution are directly dependent on water chemistry. Generally, ionic copper is more soluble in low pH waters and less soluble in high pH waters.

As an essential nutrient, copper is accumulated by plants and animals, although apparently it is not generally biomagnified. Since copper is strongly bioaccumulated, and because biogenic ligands play an important role in complexing copper, biological activity is a major factor in determining the distribution and occurrence of copper in the ecosystem. For example, bioaccumulation patterns may exhibit seasonal variations related to biological activity.

Because many copper compounds and complexes are readily soluble, copper is among the more mobile heavy metals in soil and other surface environments. The major process that limits the environmental mobility of copper is adsorption to organic matter, clays, and other materials. Atmospheric transport of copper compounds can also occur.

Health Effects

Copper appears to increase the mutagenic activity of triose reductone and ascorbic acid in bacterial test systems. However, copper itself does not appear to have mutagenic effects in animals or humans. Copper does not appear to produce teratogenic or carcinogenic effects in animals or humans. Dietary levels of trace elements such as molybdenum, sulphur, zinc, and iron can affect the level of copper that produces certain deficiency or toxicity symptoms. In general, more attention is given to the problems associated with copper deficiency than to problems of excess copper in the environment. However, high levels of copper can be toxic to humans.

Metallic copper dust exposure can cause a short-term illness similar to metal fume fever that is characterized by chills, fever, aching muscles, dryness of mouth and throat, and headache. Exposure to copper fumes can produce upper respiratory tract irritation, a metallic or sweet taste, nausea, metal fume fever, and sometimes discoloration of skin and hair. Individuals exposed to dusts and mists of copper salts may exhibit congestion of nasal mucous membranes, sometimes of the pharynx, and ocassionally ulceration with perforation of the nasal septum.

If sufficent concentrations of copper salts reach the gastrointestinal tract, they act as irritants and can produce salivation, nausea, vomiting, gastritis, and diarrhea. Elimination of ingested ionic copper by vomiting and diarrhea generally protects the patient from more serious systemic toxic effects which can include hemolysis, hepatic necrosis, gastrointestinal bleeding, oliguria, azotemia, hemoglobinuria, hematuria, proteinuria, hypotension, tachycardia, convulsions, and death. Chronic exposure may result in anemia.

Copper salts act as skin irritants producing an itching eczema. Conjunctivitis or even ulceration and turbidity of the cornea may result from direct contact of ionic copper with the eye.

Toxicity to Wildlife and Domestic Animals

Mean acute toxicity values for a large number of freshwater animals range from 7.2 µg/liter for Daphnia pulicaria to 10,200 µg/liter for the bluegill. Toxicity tends to decrease as hardness, alkalinity, and total organic carbon increase. Chronic values for a variety of freshwater species range from 3.9 µg/liter for brook trout to 60.4 µg/liter for northern pike. Hardness does not appear to affect chronic toxicity. The acute-chronic ratios for different species range from 3 to 156. The more sensitive species tend to have lower ratios than the less sensitive species. In addition, the ratio seems to increase with hardness. Acute toxicity values for saltwater organisms range from 17 µg/liter for a calanoid copepod to 600 µg/liter for the shore crab. A chronic value of 54 µg/liter and an acute-chronic ratio of 3.4 is reported for the mysid shrimp. Long term exposure to 5 µg/liter is fatal to the bay scallop.

Bioconcentration factors in freshwater species range from zero for the bluegill to 2,000 for the alga Chlorella regularis. Among saltwater species the highest bioaccumulation factors are those for the bivalve molluscs. Oysters can bioaccumulate copper up to 28,200 times without any significant mortality.

Sheep are very susceptible to copper toxicosis and poisoning may be acute or chronic. Acute poisoning is caused by direct

action of copper salts on the gastrointestinal [illegible] in gastroenteritis, shock, and death. The toxic dose is about 200 mg/kg and is usually obtained through an accidental overdose of an antihelminthic. Ingestion of excess copper over a long period of time results in absorption and accumulation of copper by the liver. This type of chronic cumulative poisoning may suddenly develop into an acute hemolytic crisis. Copper intake of 1.5 g/day for 30 days is known to be fatal for many breeds of sheep. Excessive copper may be stored in the liver as a result of excess copper ingestion, as a consequence of impaired liver function, or in connection with a deficiency or excess of other trace elements. Sheep eliminate accumulated copper very slowly after cessation of exposure.

Swine develop copper poisoning at levels of 250 mg/kg in the diet unless zinc and iron levels are increased. Toxicosis develops with hypochromic microcytic anemia, jaundice, and marked increases in liver and serum copper levels as well as serum aspartate amino transferase. High copper levels may be found in swine because of the practice of feeding high copper diets in order to increase daily weight gain. However, swine rapidly eliminate copper once it is removed from the diet. Cattle are much more resistant to copper in the diet than sheep or swine. Copper toxicity in ruminants can be counteracted by inclusion of molybdenum and sulfate in the diet.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed)

Freshwater

Acute toxicity: $e^{(0.905\ [\ln(\text{hardness})] - 1.413)}$ µg/liter

Chronic toxicity: $e^{(0.905\ [\ln(\text{hardness})] - 1.785)}$ µg/liter

Saltwater

Acute toxicity: 3.2 µg/liter
Chronic toxicity: 2.0 µg/liter

Human Health

Organoleptic criterion: 1 mg/liter

National Secondary Drinking Water Standards (USEPA): 1 mg/liter

OSHA Standards: 1.0 mg/m^3 TWA (dust and mist)
0.1 mg/m^3 TWA (fume)

ACGIH Threshold Limit Values: 1.0 mg/m^3 TWA (dusts and mists)
0.2 mg/m^3 TWA (fume)
2.0 mg/m^3 STEL (dusts and mists)

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

BOSTWICK, J.L. 1982. Copper toxicosis in sheep. J. Am. Vet. Med. Assoc. 180:386-387

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

UNDERWOOD, E.J. 1979. Trace metals in humans and animal health. J. Hum. Nutr. 35:37-48

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Copper. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-036

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Water quality criteria: Request for comments. Fed. Reg. 49:4551-4553

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Copper. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H025 (Final Draft)

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2,332 pages

CADMIUM

Summary

Cadmium is a metal that can be present in a variety of chemical forms in wastes or in the environment. Some forms are insoluble in water, but cadmium is relatively mobile in the aquatic environment. Cadmium is carcinogenic in animals exposed by inhalation and may also be in humans. It is uncertain whether it is carcinogenic in animals or humans exposed via ingestion. Cadmium is a known animal teratogen and reproductive toxin. It has chronic effects on the kidney, and background levels of human exposure are thought to provide only a relatively small margin of safety for these effects. The EPA Ambient Water Quality Criterion for protection of human health is 10 µg/liter.

Background Information

Cadmium is a soft, bluish white metal that is obtained as a by-product from the treatment of the ores of copper, lead, and iron. Cadmium has a valence of +2 and has properties similar to those of zinc. Cadmium forms both organic and inorganic compounds. Cadmium sulfate is the most common salt.

CAS Number: 7440-43-9

Chemical Formula: Cd

IUPAC Name: Cadmium

Chemical and Physical Properties

Atomic Weight: 112.41

Boiling Point: 765°C

Melting Point: 321°C

Specific Gravity: 8.642

Solubility in Water: Salts are water soluble; metal is insoluble

Solubility in Organics: Variable, based on compound

Vapor Pressure: 1 mm Hg at 394°C

Transport and Fate

Cadmium is relatively mobile in the aquatic environment compared to other heavy metals. It is removed from aqueous media by complexing with organic materials and subsequently being adsorbed to the sediment. It appears that cadmium moves slowly through soil, but only limited information on soil transport is available. Cadmium uptake by plants is not a significant mechanism for depletion of soil accumulations but may be significant for human exposure.

Health Effects

There is suggestive evidence linking cadmium with cancer of the prostate in humans. In animal studies, exposure to cadmium by inhalation caused lung tumors in rats, and exposure by injection produced injection-site sarcomas and/or Leydig-cell tumors. An increased incidence of tumors has not been seen in animals exposed to cadmium orally, but four of the five available studies were inadequate by current standards.

The evidence from a large number of studies on the mutagenicity of cadmium is equivocal, and it has been hypothesized that cadmium is not directly mutagenic but impedes repair. Cadmium is a known animal teratogen and reproductive toxin. It has been shown to cause renal dysfunction in both humans and animals. Other toxic effects attributed to cadmium include immunosuppression (in animals), anemia (in humans), pulmonary disease (in humans), possible effects on the endocrine system, defects in sensory function, and bone damage. The oral LD_{50} in the rat was 225 mg/kg.

Toxicity to Wildlife and Domestic Animals

Laboratory experiments suggest that cadmium may have adverse effects on reproduction in fish at levels present in lightly to moderately polluted waters.

The acute LC_{50} for freshwater fish and invertebrates generally ranged from 100 to 1,000 µg/liter; salmonids are much more sensitive than other organisms. Saltwater species were in general 10-fold more tolerant to the acute effects of cadmium. Chronic tests have been performed and show that cadmium has cumulative toxicity and acute-chronic ratios that range of from 66 to 431. Bioconcentration factors were generally less than 1,000 but were as high as 10,000 for some freshwater fish species.

No adverse effects on domestic or wild animals were reported in the studies reviewed.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed 1984)

Freshwater

Acute toxicity: $e^{(1.30[\ln(\text{hardness})] - 3.92)}$ µg/liter

Chronic toxicity: $e^{(0.87[\ln(\text{hardness})] - 4.38)}$ µg/liter

Saltwater

Acute toxicity: 38 µg/liter
Chronic toxicity: 12 µg/liter

Human Health

Criterion: 10 µg/liter

CAG Unit Risk for inhalation exposure (USEPA): 7.8 $(\text{mg/kg/day})^{-1}$

Interim Primary Drinking Water Standard (USEPA): 10 µg/liter

NIOSH Recommended Standards: 40 $\mu g/m^3$ TWA
200 $\mu g/m^3$/15 min Ceiling Level.

OSHA Standards: 200 $\mu g/m^3$ TWA
600 $\mu g/m^3$ Ceiling Level

ACGIH Threshold Limit Values: 50 $\mu g/m^3$ TWA

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

CLEMENT ASSOCIATES, INC. 1983. Assessment of the Weight of Evidence for Risk Assessment for Four Selected Toxic Air Pollutants. Report Prepared for the Air Economic Branch, OPRM, U.S. Environmental Protection Agency. May 1983.

FLEISCHER, M., SAROFIM, A.F., FASSETT, D.W., HAMMOND, P., SCHARKETTE, H.T., NISBET, I.C.T., and EPSTEIN, S. 1974. Environmental impact of cadmium: A review by the panel on hazardous trace substances. Environ. Health Perspect. 7:253-323

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

TAKENAKA, S., OLDIGES, H., KONIG, H., HOCHRAINER, D., and OBERDORSTER, G. 1983. Carcinogenicity of cadmium chloride aerosols in W rats. JNCI 70:367-371

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Cadmium. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-025

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Cadmium. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H038 (Final Draft)

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1985. Health Assessment Document for Dichloromethane (Methylene Chloride). Office of Health and Environmental Assessment. Washington, D.C. February 1985. EPA 600/8-82/004F

MAGNESIUM

Summary

Exposure to magnesium oxide fumes can cause metal fume fever in humans. Exposure to magnesium oxide dust can irritate the eyes and respiratory tract. Ingestion of very high levels of magnesium salts can cause central nervous sytem effects; it can also have a laxative action.

Background Information

Magnesium is the eighth most abundant element on earth. It is very reactive chemically and does not occur uncombined in nature. Finely divided magnesium can react with water to yield hydrogen gas and magnesium hydroxide. However, reaction of solid magnesium with water is self-limiting because of the formation of a film of magnesium hydroxide. As a result, elemental magnesium is considered insoluble in water.

CAS Number: 7439-95-4

Chemical Formula: Mg

IUPAC Name: Magnesium

Chemical and Physical Properties

Atomic Weight: 24.312

Boiling Point: 1107°C

Melting Point: 648.8°C

Specific Gravity: 1.738

Solubility in Water: Insoluble; most salts are very soluble

Transport and Fate

Most magnesium salts are very soluble at pH levels normally found in natural waters, and the magnesium ion is readily transported in surface water, soil, and groundwater. The extent of magnesium transport in soil is dependent, in part, on the cation exchange capacity of the soil. Evaporation of ocean spray particles and subsequent atmospheric transport of magnesium

can occur. Atmospheric transport of dusts and fumes of compounds such as magnesim oxide can also occur.

Health Effects

There is no evidence to suggest that magnesium has carcinogenic, mutagenic, teratogenic, or reproductive effects in humans or experimental animals. Magnesium oxide fumes can produce metal fume fever in humans and experimental animals. Exposure to magnesium oxide dust may cause irritation of the eyes and respiratory tract. Human exposure to magnesium usually occurs by ingestion. Magnesium is an essential element for humans, animals, and plants. Ingestion of 3.6 to 4.2 mg/kg/day is thought to be adequate for maintenance of magnesium balance in humans. The average adult American is estimated to ingest 240 to 480 mg/kg/day in food and water. However, magnesium is absorbed relatively poorly by the gastrointestinal tract and also is readily excreted in the urine. Excessive magnesium retention in the body generally only occurs as a result of severe kidney disease. Symptoms of hypermagnesemia can include a sharp drop in blood pressure, and respiratory paralysis due to central nervous system depression. Ingestion of magnesium salts at concentrations over 700 mg/liter can have a laxative effect. However, humans can adapt to ingestion of these levels in a relatively short time. Magnesium has a very unpleasant taste in water at concentrations producing toxic effects.

Different magnesium compounds vary in the severity of their toxic effects to experimental animals. Such effects include central nervous system and purgative effects similar to those seen in humans. Subcutaneous injection of powdered magnesium or magnesium alloys can produce symptoms in experimental animals resembling gas gangrene. Application of powdered magnesium to abraded skin can produce an inflammatory reaction. However, these types of skin effects have not been reported in exposed workers.

Toxicity to Wildlife and Domestic Animals

Available data are not adequate to characterize the toxicity of magnesium to wildlife or domestic animals. Observed effects are generally related to deficiency symptoms.

Regulations and Standards

OSHA Standard: 15 mg/m^3 (magnesium oxide fume)

ACGIH Threshold Limit Values:

10 mg/m^3 TWA (magnesium oxide fume)
10 mg/m^3 TWA (magnesite, nuisance particulate)
20 mg/m^3 STEL (magnesite, nuisance particulate)

U.S. Department of Transportation: Flammable solid; dangerous when wet

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

CLAYTON, G.D., and CLAYTON, F.E., eds. 1981. Patty's Industrial Hygiene and Toxicology. Vol. 2A: Toxicology. 3rd rev. ed. John Wiley and Sons, New York. 2,878 pages

DOULL, J., KLAASSEN, C.D., and AMDUR, M.O., eds. 1980. Casarett and Doull's Toxicology: The Basic Science of Poisons. 2nd ed. Macmillan Publishing Co., New York. 778 pages

NATIONAL ACADEMY OF SCIENCES (NAS). 1977. Drinking Water and Health. Safe Drinking Water Committee, Washington, D.C. 939 pages

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1984. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. April 1984

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2332 pages

MANGANESE

Summary

Manganese chloride produced lymphomas and manganese sulfate, tumors after injestion into mice. In humans, chronic exposure to manganese causes degenerative changes in the central nervous system in the form of a Parkinson-like disease; liver changes also occur. Acute exposure causes manganese pneumonitis.

CAS Number: 7439-96-5

Chemical Formula: Mn

IUPAC Name: Manganese

Chemical and Physical Properties

Atomic Weight: 54.938

Boiling Point: 1962°C

Melting Point: 1244°C

Specific Gravity: 7.20

Solubility in Water: Decomposes; some compounds are soluble

Transport and Fate

Manganese occurs most commonly in the +2 and +4 oxidation states in aquatic systems. Its solubility depends to a great extent on pH, dissolved oxygen, and presence of complexing agents. In saltwater, it is estimated that 85% or more of the manganese present exists in a soluble form. In freshwater, manganese can occur as the soluble ion, in complex organic ions, or in colloidal suspensions. Manganese often occurs at higher concentrations near the bottom of stratified lakes because it can be released from sediments, as the manganous ion, under reducing conditions.

In the soil, the concentration and chemical form in which manganese occur can be affected by pH, cation exchange capacity, drainage, organic matter content, and other factors. The solubility of manganese is increased at low pH and under reducing conditions. The presence of high concentrations of chlorides, nitrates, or sulfates may also increase solubility. Under these conditions, manganese is more easily taken up by plants

or transported in aqueous solution. Lack of sufficient cation exchange sites, which are provided by organic matter or clay, can also result in greater leaching of manganese to surface or groundwater.

Atmospheric transport of manganese fumes or dusts can occur. These materials can be returned to the earth by wet or dry deposition.

Health Effects

There are no epidemiological studies suggesting that manganese or its compounds are carcinogenic or have teratogenic or reproductive effects in humans. Exposure to manganese chloride by intraperitoneal or subcutaneous routes was reported to cause lymphomas in mice. Manganese sulfate was found to produce tumors after intraperitoneal administration in mice. No other reports of unequivocal carcinogenic activity are available for common manganese compounds. Some manganese compounds, notably manganese chloride, have exhibited mutagenic activity in a variety of test systems. Manganese compounds do not appear to be teratogenic, however.

In humans, manganese dusts and compounds have relatively low oral and dermal toxicity, but they can cause a variety of toxic effects after inhalation exposure. Acute exposure to very high concentrations can cause manganese pneumonitis, increased susceptibility to respiratory disease, and pathologic changes including epithelial necrosis and mononuclear proliferation. Chronic manganese poisoning is more common, but generally occurs only among persons occupationally exposed to manganese compounds. Degenerative changes in the central nervous system are the major toxic effects. Early symptoms include emotional changes, followed by a masklike face, retropulsion or propulsion, and a Parkinson's-like syndrome. Liver changes are also frequently seen. Individuals with an iron deficiency may be more susceptible to chronic poisoning.

Duplication of human exposure symptoms in experimental animals has only been partially successful. In rabbits exposed by inhalation to manganese dust, manganese pneumonitis did not develop, but fibrotic changes in the lungs were observed. Central nervous system effects characteristic of chronic exposure in humans have only been reproduced in monkeys.

Toxicity to Wildlife and Domestic Animals

Adequate data for characterization of the toxicity of manganese to wildlife or domestic animals are not available.

A 48-hour LC_{50} value of 16 mg/liter of manganese is reported for embryos of the oyster Crassostrea virginica. For the softshell clam Mya arenaria a 168-hour LC_{50} value of 300 mg/liter is reported.

Regulations and Standards

OSHA Standard: 5 mg/m^3 Ceiling Level

ACGIH Threshold Limit Values:

1 mg/m^3 TWA (fume)
3 mg/m^3 STEL (fume)
5 mg/m^3 Ceiling Level (dust and compounds)

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

DOULL, J., KLAASSEN, C.D., and AMDUR, M.O., eds. 1980. Casarett and Doull's Toxicology: The Basic Science of Poisons. 2nd ed. Macmillan Publishing Co., New York. 778 pages

EISLER, R. 1977. Acute toxicities of selected heavy metals to the softshell clam, Mya arenaria. Bull. Environ. Contam. Toxicol. 17:137-145

NATIONAL ACADEMY OF SCIENCE (NAS). 1973. Medical and Biological Effects of Environmental Pollutants: Manganese. Washington, D.C. 191 pages

NATIONAL ACADEMY OF SCIENCES (NAS). 1977. Drinking Water and Health. Safe Drinking Water Committee, Washington, D.C. 939 pages

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Manganese. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H057 (Final Draft)

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2332 pages

CHROMIUM

Summary

Chromium is a heavy metal that generally exists in either a trivalent or hexavalent oxidation state. Hexavalent chromium (Cr VI) is rather soluble and is quite mobile in groundwater and surface water. However, in the presence of reducing agents it is rapidly converted to trivalent chromium (Cr III), which is strongly adsorbed to soil components and consequently is much less mobile. A number of salts of hexavalent chromium are carcinogenic in rats. In addition, an increased incidence of lung cancer was seen in workers occupationally exposed to chromium VI. Hexavalent chromium also causes kidney damage in animals and humans. Trivalent chromium is less toxic than hexavalent chromium; its main effect is contact dermatitis in sensitive individuals. The EPA Ambient Water Quality Criteria for the protection of human health are 50 µg/liter for Cr VI and 170 µg/liter for Cr III.

CAS Number: 7440-47-3

Chemical Formula: Cr

IUPAC Name: Chromium

Chemical and Physical Properties (Metal)

Atomic Weight: 51.996

Boiling Point: 2672°C

Melting Point: 1857 ± 20°C

Specific Gravity: 7.20 at 28°C

Solubility in Water: Insoluble; some compounds are soluble

Transport and Fate

Hexavalent Cr is quite soluble, existing in solution as a component of a complex anion. It is not sorbed to any significant degree by clays or hydrous metal oxides. The anionic form varies according to pH and may be a chromate, hydrochromate, or dichromate. Because all anionic forms are so soluble, they are quite mobile in the aquatic environment. Cr VI is efficiently

removed by activated carbon and thus may have some affinity for organic materials in natural water. Cr VI is a moderately strong oxidizing agent and reacts with reducing materials to form trivalent chromium. Most Cr III in the aquatic environment is hydrolyzed and precipitates as chromium hydroxide. Sorption to sediments and bioaccumulation will remove much of the remaining Cr III from solution. Cr III is adsorbed only weakly to inorganic materials. Cr III and Cr VI are readily interconvertible in nature depending on microenvironmental conditions such as pH, hardness, and the types of other compounds present. Soluble forms of chromium accumulate if ambient conditions favor Cr VI. Conditions favorable for conversion to Cr III lead to precipitation and adsorption of chromium in sediments.

In air, chromium is associated almost entirely with particulate matter. Sources of chromium in air include windblown soil and particulate emissions from industrial processes. Little information is available concerning the relative amounts of Cr III and Cr VI in various aerosols. Relatively small particles can form stable aerosols and can be transported many miles before settling out.

Cr III tends to be adsorbed strongly onto clay particles and organic particulate matter, but can be mobilized if it is complexed with organic molecules. Cr III present in minerals is mobilized to different extents depending on the weatherability and solubility of the mineral in which it is contained. Hexavalent compounds are not strongly adsorbed by soil components and Cr VI is mobile in groundwater. Cr VI is quickly reduced to CR III in poorly drained soils having a high content of organic matter. Cr VI of natural origin is rarely found in soils.

Health Effects

The hexavalent form of chromium is of major toxicological importance in higher organisms. A variety of chromate (Cr VI) salts are carcinogenic in rats and an excess of lung cancer has been observed among workers in the chromate-producing industry. Cr VI compounds can cause DNA and chromsome damage in animals and humans, and Cr (VI) trioxide is teratogenic in the hamster. Inhalation of hexavalent chromium salts causes irritation and inflammation of the nasal mucosa, and ulceration and perforation of the nasal septum. Cr VI also produces kidney damage in animals and humans. The liver is also sensitive to the toxic effects of hexavalent Cr, but apparently less so than the kidneys or respiratory system. Cr III is less toxic than Cr VI; its main effect in humans is a form of contact dermatitis in sensitive individuals.

Cr III:

Aquatic Life (Proposed Criteria)

Freshwater

Acute toxicity: $e^{(0.819[\ln(\text{hardness})]+3.568)}$ µg/liter

Chronic toxicity: $e^{(0.819\ [\ln(\text{hardness})])+0.537)}$ µg/liter

Saltwater

The available data are not adequate for establishing criteria.

Human Health

Criterion: 170 µg/liter

CAG Unit Risk for inhalation exposure to CR VI (USEPA): 41 $(\text{mg/kg/day})^{-1}$

National Interim Primary Drinking Water Standard: 50 µg/liter

NIOSH Recommended Standards for CR VI: 1 $\mu g/m^3$ carcinogenic
25 $\mu g/m^3$ noncarcinogenic TWA
50 $\mu g/m^3$ noncarcinogenic (15-min sample)

OSHA Standards: OSHA air standards have been set for several chromium compounds. Most recognized or suspected carcinogenic chromium compounds have ceiling limits of 100 $\mu g/m^3$.

ACGIH Threshold Limit Values: Several chromium compounds have TWAs ranging from 0.05 to 0.5 mg/m^3. Chromite ore processing (chromate), certain water insoluble Cr VI compounds, and chromates of lead and zinc are recognized or suspected human carcinogens and have 0.05 mg/m^3 TWAs.

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

INTERNATIONAL AGENCY FOR RESEARCH ON CANCER (IARC). 1980. IARC Monograph on the Evaluation of the Carcinogenic Risk of Chemicals to Humans. Vol. 23: Some Metals and Metallic Compounds. World Health Organization, Lyon, France

Cyanides move rather freely in soils but biodegradation would probably significantly decrease the amount present in the groundwater. Volatilization of HCN and nitriles may occur from soil surfaces.

Health Effects

Hydrogen cyanide and its simple salts, such as sodium cyanide, are highly toxic by all routes. Many reports are available regarding acute poisoning in humans. Hydrogen cyanide vapor is irritating at very low concentrations, is considered dangerous at 20 ppm (20 mg/m^3), and is fatal at concentrations of 100 ppm (100 mg/m^3) for one hour. NIOSH notes reports of chronic poisoning resulting in fatigue, weariness and other subjective symptoms in workers, but these findings have been disputed by other investigators. Chronic exposure to low levels of cyanide salts has been reported to cause enlargement of the thyroid gland in humans, apparently due to inefficient elimination of the cyanide metabolite thiocyanate. NIOSH (1976) concluded that there was no evidence of carcinogenicity, mutagenicity, or teratogenicity for cyanides. Cyanide has been shown to produce chromosome breaks in a plant, Vicia faba. Because of its mechanism of action, inhibition of the election transport system in oxidative phosphorylation, cyanide is acutely toxic to almost all forms of life. A reduction in the TLV for HCN from 10 mg/m^3 to a ceiling value of 3 mg/m^3 has been recommended by several investigators, to prevent the various nonspecific effects noted by several investigators (ACGIH 1980).

Toxicity to Wildlife and Domestic Animals

Cyanide is acutely toxic to both freshwater and saltwater organisms, causing death at levels of about 50 µg/liter in sensitive species and being fatal to many species at levels above 200 µg/liter. Final acute values were determined to be 44.7 µg/liter for freshwater species and 2.03 µg/liter for saltwater species. Effects such as reduced survival and reduced reproduction were seen in fish chronically exposed to free cyanide concentrations of from 10 to 50 µg/liter. The final acute chronic ratios were determined to be 10.7 and 3.5 for freshwater and saltwater organisms, respectively. The final chronic values were determined by dividing the acute values by the acute-chronic ratio, and were determined to be 4.2 for freshwater species and 0.57 for saltwater organisms. An accidental spill of cyanide caused the death of 4,800 fish in Oak Ridge, Tennessee. The long-term effects of this spill were not reported. Livestock death and environmental damage were caused by high levels of cyanide leaching from a drum disposal site in Illinois.

Summary

Cyanide can be present in many forms in the environment. The transport, fate, and toxicity of the chemical is quite dependent on the specific form. Hydrogen cyanide and its simple salts are highly toxic following acute exposure by humans, experimental animals, and both aquatic and terrestrial wildlife.

Background Information

Cyanide (CN-) is usually defined as hydrogen cyanide (HCN) and its salts. The chemical/physical properties, transport and fate, and toxicity of cyanide are quite dependent on the form of cyanide present.

CAS Number: 151-50-8; 143-33-9

Chemical Formula: CN-

IUPAC Name: Cyanide

Chemical and Physical Properties

Molecular Weight: 27 (HCN)

Boiling Point: 26.7°C (HCN)

Melting Point: -14°C (HCN)

Specific Gravity: 0.699 at 22°C (HCN)

Solubility in Water: Soluble (HCN)

Solubility in Organics: Soluble in alcohol and ether

Vapor Pressure: 657.8 mm Hg at 21.9°C (HCN)

Transport and Fate

The transport and fate of cyanide in the environment is dependent on the chemical compound containing the cyanide. Most free cyanide will be HCN in aquatic environments and will probably evaporate, although biodegradation is another possible fate process. Metal cyanides are generally insoluble and for

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed)

Freshwater

Acute toxicity: 22 µg/liter
Chronic toxicity: 4.2 µg/liter

Saltwater

Acute toxicity: 1.0 µg/liter
Chronic toxicity: 0.57 µg/liter

Human Health

Criterion: 200 µg/liter

Primary Drinking Water Standard (USEPA): 200 µg/liter

ACGIH Threshold Limit Value: 5 mg/m^3 TWA

REFERENCES

AMERICAN COUNCIL OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1976. Criteria for a Recommended Standard--Occupational Exposure to Hydrogen Cyanide and Cyanide Salts (NaCN, KCN, and $Ca(CN)_2$). Washington, D.C. DHEW Publication No. (NIOSH) 77-108

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Cyanides. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-037

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1983. Section B of Ambient Water Criteria for Cyanide--Aquatic Toxicology. Draft Report. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. August 1983

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Cyanide. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H011 (Final Draft)

VERSCHUEREN, K. 1977. Handbook of Environmental Data on Organic Chemicals. Van Nostrand Reinhold Co., New York. 659 pages

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2,332 pages

Summary

In the absence of photolytic degradation, dimethylnitrosamine is probably persistent in the environment. Dimethylnitrosamine is carcinogenic and produces lung, liver, and kidney tumors in rats and mice and liver tumors in several other animal species. It also exhibits transplacental carcinogenicity in animals and is mutagenic and embryotoxic. Both acute and chronic exposure have adverse effects on the liver in humans and experimental animals.

CAS Number: 62-75-9

Chemical Formula: $(CH_3)_2NNO$

IUPAC Name: n-Nitrosodimethylamine

Important Synonyms and Trade Names: n-Methyl-n-nitrosomethanamine, n,n-dimethylnitrosamine, DMN, DMNA, NDMA

Chemical and Physical Properties

Molecular Weight: 74.1

Boiling Point: 151°C

Specific Gravity: 1.0 at 20°C

Solubility in Water: Soluble in all proportions

Solubility in Organics: Soluble in organic solvents, lipids

Log Octanol/Water Partition Coefficient: 0.06 to -0.69

Transport and Fate

The most probable environmental fate of dimethylnitrosamine in aqueous solution appears to be slow photolytic degradation. Furthermore, although supporting data are limited, it has been speculated that hydrogen bonding of dimethylnitrosamine with humic acids or coordination with metal cations produces a photolabile intermediate and could lead to moderately rapid degradation in surface waters. Dimethylnitrosamine has been detected

in the atmosphere of metropolitan areas and near manufacturing facilities emitting this compound, suggesting that some atmospheric transport can occur. However, it is reported that photolytic degradation in air would be rapid, with a half-life of less than 1 hour. Airborne concentrations in excess of a few parts per billion appear to be unlikely except near sources of direct emissions. There is no evidence to suggest that oxidation or hydrolysis are important environmental fates.

Dimethylnitrosamine is completely miscible in water and is reported to be highly solvated. This information, along with limited experimental data, suggest that volatilization from surface waters is probably not an important process. Dimethylnitrosamine has a log octanol/water partition coefficient near 0; significant sorption by organic particulates is therefore unlikely. Experimental evidence confirms this and further suggests that sorption by clay particulates in wet soil is also unlikely. Because dimethylnitrosamine is completely miscible in water and has a low log octanol/water partition coefficient, bioaccumulation is probably an insignificant process. Although biodegradation in surface waters does not appear to be an important environmental fate, slow degradation of dimethylnitrosamine in sewage and soil is reported to occur. Based on this information, it is likely that in the absence of photolytic degradation dimethylnitrosamine would be very persistent in the environment.

Health Effects

Dimethylnitrosamine is considered to be carcinogenic in many experimental animal species by various routes of exposure. Dose-response relationships have been established in several studies. This compound produces liver, lung, and kidney tumors in some species of mice and rats after oral and inhalation exposure. Increased incidences of liver tumors have also been observed in many other animal species after oral administration. Inhalation exposure in rats has produced tumors of the ethroturbinals and nasal cavity. Although insufficient epidemiologic evidence exists to establish a causative role for dimethylnitrosamine in human carcinogenesis, IARC and other public health organizations recommend that this compound be regarded as a human carcinogen.

Dimethylnitrosamine is mutagenic in many microbial test systems with metabolic activation and in several other in vivo and in vitro test systems. This compound is reported to exhibit transplacental carcinogenicity and to be embryotoxic. No teratogenic effects have been reported. Acute and chronic exposure of humans and experimental animals to dimethylnitrosamine resulted primarily in a variety of hepatotoxic effects. In rats, an oral LD_{50} value of 40 mg/kg and an inhalation LD_{50} value of 37 mg/kg are reported.

Toxicity to Wildlife and Domestic Animals

In crayfish exposed to dimethylnitrosamine in water for 6 months, extensive antennal gland degeneration was observed at 200,000 µg/liter and hyperplasia of hepatopancreas tubular cells at 100,000 µg/liter. Rainbow trout fed dimethylnitrosamine for 52 weeks showed a dose-related increase in hepatocellular carcinoma at doses of 200, 400, and 800 mg/kg. The weighted average bioconcentration factor for the edible portion of all freshwater and estuarine aquatic organisms consumed by Americans is 0.026.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life

Freshwater

Acute toxicity: The available data for nitrosamines in general indicate that toxic effects occur at concentrations as low as 5,850 µg/liter and would occur at lower concentrations among species that are more sensitive than those tested.

Chronic toxicity: No available data

Saltwater

Acute toxicity: The available data for nitrosamines in general indicate that toxic effects occur at concentrations as low as 3,300,000 µg/liter and would occur at lower concentrations among species that are more sensitive than those tested.

Chronic toxicity: No available data

Human Health

Estimates of the carcinogenic risks associated with lifetime exposure to various levels of dimethylnitrosamine in water are:

Risk	Concentration
10^{-5}	14 ng/liter
10^{-6}	1.4 ng/liter
10^{-7}	0.14 ng/liter

ACGIH Threshold Limit Value: Suspected human carcinogen

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

INTERNATIONAL AGENCY FOR RESEARCH ON CANCER (IARC). IARC Monographs on the Evaluation of Carcinogenic Risk of Chemicals to Humans. Vol. 17: Some N-Nitroso Compounds. World Health Organization, Lyon, France. Pp. 125-175

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1984. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1984

SAX, N.I. 1975. Dangerous Properties of Industrial Materials. 4th ed. Van Nostrand Reinhold Co., New York. 1,258 pages

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Nitrosamines. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-064

Solubility in Organics: Depends on chemical species

Vapor Pressure: 0.0012 mm Hg at 20°C

Transport and Fate

Mercury and certain of its compounds, including several inorganic species and dimethyl mercury, can volatilize to the atmosphere from aquatic and terrestrial sources. Volatilization is reduced by conversion of metallic mercury to complexed species and by deposition of HgS in reducing sediments, but even so atmospheric transport is the major environmental distribution pathway for mercury. Precipitation is the primary mechanism for removal of mercury from the atmosphere. Photolysis is important in the breakdown of airborne mercurials and may be important in some aquatic systems. Adsorption onto suspended and bed sediments is probably the most important process determining the fate of mercury in the aquatic environment. Sorption is strongest into organic materials. Mercury in soils is generally complexed to organic compounds.

Virtually any mercury compound can be remobilized in aquatic systems by microbial conversion to methyl and dimethyl forms. Conditions reported to enhance biomethylation include large amounts of available mercury, large numbers of bacteria, the absence of strong complexing agents, near neutral pH, high temperatures, and moderately aerobic environments. Mercury is strongly bioaccumulated by numerous mechanisms. Methylmercury is the most readily accumulated and retained form of mercury in aquatic biota, and once it enters a biological system it is very difficult to eliminate.

Health Effects

When administered by intraperitoneal injection, metallic mercury produces implantation site sarcomas in rats. No other studies were found connecting mercury exposure with carcinogenic effects in animals or humans. Several mercury compounds exhibit a variety of genotoxic effects in eukaryotes. In general, organic mercury compounds are more toxic than inorganic compounds. Although brain damage due to prenatal exposure to methylmercury has occurred in human populations, no conclusive evidence is available to suggest that mercury causes anatomical defects in humans. Embryotoxicity and teratogenicity of methylmercury has been reported for a variety of experimental animals. Mercuric chloride is reported to be teratogenic in experimental animals. No conclusive results concerning the teratogenic effects of mercury vapor are available.

MERCURY

Summary

Both organic and inorganic forms of mercury are reported to be teratogenic and embryotoxic in experimental animals. In humans, prenatal exposure to methylmercury has been associated with brain damage. Other major target organs for organic mercury compounds in humans are the central and peripheral nervous system and the kidney. In animals, toxic effects also occur in the liver, heart, gonads, pancreas, and gastrointestinal tract. Inorganic mercury is generally less acutely toxic than organic mercury compounds, but it does affect the central nervous system adversely. The EPA Ambient Water Quality Criterion for the protection of human health is 144 ng/liter.

Background Information

Several forms of mercury, including insoluble elemental mercury, inorganic species, and organic species, can exist in the environment. In general, the mercurous (+1) salts are much less soluble than the more commonly found mercuric (+2) salts. Mercury also forms many stable organic complexes that are generally much more soluble in organic liquids than in water. The nature and solubility of the chemical species that occur in an environmental system depend on the redox potential and the pH of the environment.

CAS Number: 7439-97-6

Chemical Formula: Hg

IUPAC Name: Mercury

Chemical and Physical Properties (Metal)

Atomic Weight: 200.59

Boiling Point: 356.58°C

Melting Point: -38.87°C

Specific Gravity: 13.5939 at 20°C

Solubility in Water: 81.3 µg/liter at 30°C; some salts and organic compounds are soluble

Chronic dietary exposure of chickens to mercuric chloride at growth inhibitory levels causes immune suppression, with a differential reduction effect on specific immunoglobulins.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed Criteria)

Freshwater

Acute toxicity: 1.1 µg/liter
Chronic toxicity: 0.20 µg/liter

Saltwater

Acute toxicity: 1.9 µg/liter
Chronic toxicity: 0.10 µg/liter

Human Health

Criterion: 144 ng/liter

Primary Drinking Water Standard: 0.002 mg/liter

NIOSH Recommended Standard: 0.05 mg/m^3 TWA (inorganic mercury)

OSHA Standard: 0.1 mg/m^3 Ceiling Level

ACGIH Threshold Limit Values:

0.01 mg/m^3 TWA (alkyl compounds)
0.03 mg/m^3 STEL (alkyl compounds)
0.05 mg/m^3 TWA (vapor)
0.1 mg/m^3 TWA (aryl and inorganic compounds)

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

BRIDGER, M.A., and THAXTON, J.P. 1983. Humoral immunity in the chicken as affected by mercury. Arch. Environ. Contam. Toxicol. 12:45-49

In humans, alkyl mercury compounds pass through the blood brain barrier and the placenta very rapidly, in contrast to inorganic mercury compounds. Major target organs are the central and peripheral nervous systems, and the kidney. Methylmercury is particularly hazardous because of the difficulty of eliminating it from the body. In experimental animals, organic mercury compounds can produce toxic effects in the gastrointestinal tract, pancreas, liver, heart, and gonads, with involvement of the endocrine, immunocompetent, and central nervous systems.

Elemental mercury is not highly toxic as an acute poison. However, inhalation of high concentrations of mercury vapor can cause pneumonitis, bronchitis, chest pains, dyspnea, coughing, stomatitis, gingivitis, salivation, and diarrhea. Soluble mercuric salts are highly poisonous on ingestion, with oral LD_{50} values of 20 to 60 mg/kg reported. Mercurous compounds are less toxic when administered orally. Acute exposure to mercury compounds at high concentrations causes a variety of gastrointestinal symptoms and severe anuria with uremia. Signs and symptoms associated with chronic exposure involve the central nervous system and include behavioral and neurological disturbances.

Toxicity to Wildlife and Domestic Animals

The toxicity of mercury compounds has been tested in a wide variety of aquatic organisms. Although methylmercury appears to be more toxic than inorganic mercuric salts, few acute or chronic toxicity tests have been conducted with it. Among freshwater species, the 96-hour LC_{50} values for inorganic mercuric salts range from 0.02 µg/liter for crayfish to 2,000 µg/liter for caddisfly larvae. Acute values for methylmercuric compounds and other mercury compounds are only available for fishes. In rainbow trout, methylmercuric chloride is about ten times more toxic to rainbow trout than mercuric chloride, which is acutely toxic at about 300 µg/liter at 10°C. Methylmercury is the most chronically toxic of the tested compounds, with chronic values for Daphnia magna and brook trout of 1.00 and 0.52 µg/liter, respectively. The acute-chronic ratio for Daphnia magna is 3.2.

Mean acute values for saltwater species range from 3.5 to 1,680 µg/liter. In general, molluscs and crustaceans are more sensitive than fish to the acute toxic effects of mercury. A life-cycle experiment with the mysid shrimp showed that inorganic mercury at a concentration of 1.6 µg/liter significantly influences time of appearance of first brood, time of first spawn, and productivity. The acute-chronic ratio for the mysid shrimp is 2.9.

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances Data Base. Washington, D.C. October 1983

SHEPARD, T.H. 1980. Catalog of Teratogenic Agents. 3rd ed. Johns Hopkins University Press, Baltimore. 410 pages

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Mercury. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-058

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Water quality criteria; Request for comments. Fed. Reg. 49: 4551-4553

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Mercury. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H042 (Final Draft)

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2,332 pages

WORLD HEALTH ORGANIZATION (WHO). 1976. Environmental Health Criteria: 1. Mercury. World Health Organization, Geneva. 131 pages

POLYCHLORINATED BIPHENYLS

Summary

Polychlorinated biphenyls (PCBs) are very persistent in the natural environment and are readily bioaccumulated. In humans, exposure to PCBs has been associated with chloracne, impairment of liver function, a variety of neurobehavioral symptoms, menstrual disorders, minor birth abnormalities, and an increased incidence of cancer. Experimental animals exposed to PCBs experienced an increased incidence of cancer; reproductive problems; neurobehavioral degradation; pathological changes in the liver, stomach, skin, and other organs; and suppression of immunological function. PCBs are often contaminated, and these contaminants may be much more toxic than the PCBs themselves. The EPA Ambient Water Quality Criterion for the protection of human health corresponding to an excess cancer risk of 10^{-6} is 0.079 ng/liter.

Background Information

Polychlorinated biphenyls (PCBs) are complex mixtures of chemicals composed of two connected benzene rings with 1 to 10 chlorine atoms attached. The chemical, physical, and biological properties of these materials depend to a large degree on the amount and location of the chlorine atoms on the two benzene rings of each specific PCB and on the particular mixture of individual chlorobiphenyls that comprise the mixture.

CAS Number: 1336-36-3

Chemical Formula: $C_6H_5Cl_xC_6H_5Cl_x$

IUPAC Name: Specific for each polychlorinated biphenyl

Important Synonyms and Trade Names: PCBs, chlorinated biphenyls, polychlorobiphenyls, Aroclor, Kanechlor, Clophen

Chemical and Physical Properties

Molecular Weight: 189-399*

Boiling Point: 267°C and up*

Melting Point: 54-310°C*

---

*Increases with increasing chlorination.

Specific Gravity: 1.3 to 1.5 at 20°C*

Solubility in Water: 0.003-0.6 mg/liter

Solubility in Organics: Soluble in most common organic solvents

Log Octanol/Water Partition Coefficient: 4-6*

Vapor Pressure: 10^{-3}-10^{-5} mm Hg at 20°C**

Henry's Law Constant: 10^{-3} to -10^{-5} atm m^3/mole

Transport and Fate

The transport and fate of polychlorinated biphenyls has been studied extensively, and although individual chemicals vary in the rates at which processes occur, some generalizations can be made about PCBs as a class. PCBs are relatively inert, and therefore persistent, compounds, with low vapor pressures, low water solubility, and high log octanol/water partition coefficients. Despite their low vapor pressures, they have a high activity coefficient in water, which causes a higher rate of volatilization than might normally be expected. Volatilization and persistence account for the ubiquitous nature of PCBs in the environment. Adsorption to the organic material in soil or sediments is probably the major fate of at least the more heavily chlorinated PCBs. Once bound, the PCBs may persist for years with slow desorption providing continuous, low-level exposure to the surrounding locality. Bioaccumulation of PCBs also occurs, with most of the compound stored in the adipose tissue of the body. PCBs are degraded primarily by two routes. Less heavily chlorinated PCBs (mainly the mono-, di-, and trichlorinated PCBs) can be biodegraded by some soil microorganisms. PCBs with five or more chlorines are not measurably biodegraded. These heavier PCBs can be photolyzed by ultraviolet light. This process is extremely slow, but it may be the most important degradation process for these very persistent compounds.

Assessing the toxicity of PCBs is complicated by the fact that several different mixtures have been produced and distributed commercially and by the presence of highly toxic contaminants in some commercial mixtures. Some of these contaminants can be formed by combustion of PCBs or even by high-temperature treatment in service, so that used materials may be more toxic than the commercial mixtures whose toxicity has been studied.

---

*Increases with increasing chlorination.
**Decreases with increasing chlorination.

Health Effects

In humans exposed to PCBs (in the workplace or via accidental contamination of food), reported adverse effects include chloracne (a long-lasting, disfiguring skin disease), impairment of liver function, a variety of neurobehavioral and affective symptoms, menstrual disorders, minor birth abnormalities, and probably increased incidence of cancer. Animals experimentally exposed to PCBs have shown most of the same symptoms, as well as impaired reproduction; pathological changes in the liver, stomach, skin, and other organs; and suppression of immunological functions. PCBs are carcinogenic in rats and mice and, in appropriate circumstances, enhance the effects of other carcinogens. Reproductive and neurobiological effects of PCBs have been reported in rhesus monkeys at the lowest dose level tested, 11 µg/kg body weight/day over a period of several months.

Toxicity to Wildlife and Domestic Animals

Polychlorinated biphenyls are bioaccumulated and can be biomagnified. Therefore, their toxicity increases with length of exposure and position of the exposed species on the food chain. The toxicity of the various PCB mixtures is also dependent on their composition. Because of the complexity of PCB toxicity, only general effects will be discussed here.

The 96-hour LC_{50} values for rainbow trout, bluegills, and channel catfish were around 20 mg/liter. The same species exposed for 10 to 20 days had LC_{50} values of about 0.1 mg/liter. Invertebrate species were also adversely affected, with some species having 7-day LC_{50} values as low as 1 µg/liter. In general, juvenile organisms appeared more susceptible to the effects of PCBs than either eggs or adults.

Three primary ways in which PCBs can affect terrestrial wildlife are outright mortality, adversely affecting reproduction, and changing behavior. PCB doses greater than 200 ppm in the diet or 10 mg/kg body weight (bw) caused some mortality in sensitive bird species exposed for several days. Doses around 1,500 ppm (diet) or about 100 mg/kg (bw) caused extensive mortality in these sensitive species. They generally caused some mortality in all species, with the level being dependent on the length of exposure and the particular PCB mixture. Some mammalian species are especially susceptible to PCBs. For example, mink died when fed as little as 5 ppm in the diet (equivalent to less than 1 mg/kg bw/day). PCBs caused lower egg production; deformities; decreased hatchability, growth, and survival; and some eggshell thinning in reproductive studies on chickens fed doses of 20 ppm in the diet (1 mg/kg bw). Mink fed 1 ppm in the diet (0.2 mg/kg bw) had lower reproductive success, and there are indications that an increased incidence

of premature births in some marine mammals was linked to PCB exposure. Behavioral effects on wildlife include increased activity, decreased avoidance response, and decreased nesting, all of which could significantly influence survival in the wild.

No toxic effects on domestic animals other than chickens were reported in the sources reviewed, but susceptible species would probably be affected in a similar manner to laboratory animals and wildlife.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life

Freshwater

Acute toxicity: 2 µg/liter
Chronic toxicity: 0.014 µg/liter

Saltwater

Acute toxicity: 10 µg/liter
Chronic toxicity: 0.030 µg/liter

Human Health

Estimates of the carcinogenic risks associated with lifetime exposure to various concentrations of PCBs in water are:

Risk	Concentration
10^{-5}	0.79 ng/liter
10^{-6}	0.079 ng/liter
10^{-7}	0.0079 ng/liter

CAG Unit Risk (USEPA): 4.34 $(mg/kg/day)^{-1}$

NIOSH Recommended Standard: 1.0 $\mu g/m^3$ TWA

ACGIH Threshold Limit Value: 0.5 mg/m^3 TWA

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. *Documentation of the Threshold Limit Values.* 4th ed. Cincinnati, Ohio. 438 pages

INTERNATIONAL AGENCY FOR RESEARCH ON CANCER (IARC). 1978. IARC Monographs on the Evaluation of Carcinogenic Risk of Chemicals to Humans. Vol. 18: Polychlorinated Biphenyls and Polybrominated Biphenyls. World Health Organization, Lyon, France. Pp. 43-103

NATIONAL ACADEMY OF SCIENCES (NAS). 1977. Drinking Water and Health. Safe Drinking Water Committee, Washington, D.C. 939 pages

ROBERTS, J.R., RODGERS, D.W., BAILEY, J.R., and RORKE, M.A. 1978. Polychlorinated Biphenyls: Biological Criteria for an Assessment of their Effects on Environmental Quality. National Research Council of Canada, Ottawa, Canada. NRCC No. 16077

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1976. National Conference on Polychlorinated Biphenyls (November 19-21, 1975, Chicago, Illinois). Office of Toxic Substances, Washington, D.C. March 1976. EPA 560/6-75-004

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Polychlorinated Biphenyls (PCBs). Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-054

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Polychlorinated Biphenyls. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H004 (Final Draft)

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1985. Health Assessment Document for Dichloromethane (Methylene Chloride). Office of Health and Environmental Assessment. Washington, D.C. February 1985. EPA 600/8-82/004F

FEB 18 1987

Dick Ruelle
U.S. Fish & Wildlife Service
1830 Second Avenue
Rock Island, Illinois 61201

Re: Crab Orchard Lake RI/FS

Dear Mr. Ruelle:

Attached is a tabulation of QA/QC comments on the Phase I data and recommended date useability. The QA/QC comments are taken from memos and discussions with personnel from the U.S. Environmental Protection Agency, Region V, Contract Laboratory Management Section (CPMS) and the Quality Assurance Office (QAO).

The recommended data useability is a concurrence of opinion between myself and personnel from the CPMS and QAO. If you or your contractors disagree with these recommendations, please contact me.

My understanding is that if data is considered not useable, it should not even be included in Remedial Investigation Report. If data is useable for screening only, this should be identified in the Report and the data should be separated from data that is considered quantitative. Other data qualifiers should also be identified in the Report.

It should be noted that specific additional documentation needed to validate the data are identified for the ETC CLP organics and Weston explosives. If this missing documentation is submitted, it is possible that this data will be determined to be more useable.

Sincerely,

ORIGINAL SIGNED BY
RICHARD BOICE

Richard E. Boice
Remedial Site Project Manager

cc: B. Cowles, IEPA
D. Iyer, O'Brien & Gere

Laboratory	Parameters	QA/QC Problems[1] (Reference)	Recommended Data Useability
O'Brien & Gere	CLP Organics Volatiles	No tuning data Mass ratio outside limits for brominated cmpds. Matrix spikes outside for some compounds. (2/22/86 memo from Pat Churilla)	Not useable for: 2-butanone, vinyl acetate, 4-methyl-2-pentanone. Positive values are estimates. Negatives are not useable unless nothing detected in whole fraction.
	BNA's	Tuning data unacceptable. Compounds not found in initial calibration. Calibration data missing on two dates. Low recovery on some samples. (8/11/86 memo from Patrick Churilla)	Not useable for: aniline, bis (2-chloro-isopropyl) ether, 4-chloroaniline, 2-nitro-sodiphenylamine, benzidene, 3,3-dichloro-benzidine, di-n-octyl-phthalate, benzo(a)pyrene, indeno (1,2,3) pyrene, dibenzo(ah) anthracene. Positive values are estimates, Negatives are not useable unless nothing detected in whole fraction.
	Pesticide/PCBs	Retension time shift. (4/24/86 and 10/16/86 memos from Patrick J. Churilla)	Useable for screening purposes. PCB results are qualitative. (p. 2, 9/26/86 from James Adams)
	GC screening of soils	Data not assessed in detail	Useable for screening. PCB results are qualitative. (p. 2 of 9/26/86 memo from James Adams)

Footnote:

(1) Chain-of-custody procedures were not followed in the laboratory for any Phase I samples (p. 2, 7/3/86 memo from James Adams).

Laboratory	Parameters	QA/QC Problems (Reference)	Recommended Data Useability
O'Brien & Gere	PCB's	(9/26/86 memo from James Adams)	Useable
	Atomic Absorption Screen (Ag, As, Be, Cd, Cu, Ni, Pb, Se, Zn)	No raw data and very little QA/QC documentation is available. Results appear to be inconsistent with results from previous samplings. (9/26/86 memo from James Adams)	Not useable
	Mercury	Very high blank and poor calibration (8/1/86 memo from Jay Thakkar; 9/[illegible]'86 memo from James Adams)	Not useable
	TKN, total phosphorus	Strip charts not matched with raw data. Duplicates and spikes out of control. (8/1/86 memo from Jay Thakkar)	Values estimated. Useable only for screening relative values.
	Cyanide	High detection limit. No laboratory QC performed. (8/1/86 memo from Jay Thakkar, 7/3/86 memo from James Adams)	Not useable
	Water extractable nitrate-nitrite	Some strip charts very noisy. Wrong reading from strip charts (8/1/86 memo from Jay Thakkar)	Values estimated. Useable only for screening relative values.
	Water extractable TOC, ammonia, chloride spec cond., pH, sulfate	No records reviewed.	Probably useable for screening of relative concentrations only.
	TOX in water	No records reviewed.	Probably useable for screening of relative concentrations only.
	TOX in soils	Method not determined.	Not determined.

Laboratory	Parameters	QA/QC Problems (Reference)	Recommended Data Useability
ETC	ICP metals	(8/1/86 memo from Jay Thakkar)	Estimated values. Useable for screening purposes only.
	CLP Organics	No calibration data for VOA's & BNA's. No raw data on spikes and method blanks. No duplicate spike. BNA contained TIC's that could cause false positives. (8/11/86 memo from Patrick Churilla)	PCB results useable; BNA & Volatiles are estimated and useable for screening only. Pesticide-not determined because no documentation provided.
	Dioxin/Furan	Diphenylethers not checked as interference (8/11/86 memo from Patrick Churilla)	Useable
Weston	Explosive residues	Detection limits should be 5x higher. No retention time windows. Percent moisture not reported. Method of calculation not given. Operating conditions not reported. (7/16/86 memo from Patrick Churilla)	Useable with detection limit 5 x detection limit reported.
Hazelton	CLP Organics	(1/14/87 memo from Patrick Churilla)	Volatiles, BNA & PCB's useable except that where acetone, methylene chloride or phthalates detected in blank, there may be false positives and other positive values will be estimated and biased high. Pesticide analyses should be considered qualitative.
	ICAP metals	Not assessed	

Laboratory	Parameters	QA/QC Problems (Reference)	Recommended Data Usability
	Cyanide	Not assessed	
	Mercury	Not assessed	
	TKN, ammonia, phosphorus	Not assessed	
Upstate Laboratories	Arsenic, selenium	Not assessed	

22:23

BIOLOGICAL DATA RELEVANT TO THE EVALUATION OF CARCINOGENIC RISK TO HUMANS

Prepared for
Scientific Advisory Panel,
Safe Drinking Water and Toxic Enforcement Act
State of California

Prepared by
Dr. Raymond D. Harbison
Dr. Robert C. James
Dr. Stephen M. Roberts

Division of Interdisciplinary Toxicology
University of Arkansas
School of Medicine
Little Rock, Arkansas
U.S.A.

August 1987

TABLE OF CONTENTS

EXECUTIVE SUMMARY

Evaluation of Animal Data

The investigation of PCBs' carcinogenic potential in mice is limited to two short studies, while some eight to ten studies have been reported using various strains of rat. The PCB mixtures tested thus far are : 1) in mice - Kanechlor 300, Kanechlor 400, Kanechlor 500 and Aroclor 1254; and 2) in rats - Kanechlor 300, Kanechlor 400, Kanechlor 500, Clophen A30, Clophen A60, Aroclor 1254 and Aroclor 1260.

PCBs have been found to be tumorigenic in mice with Aroclor 1254 producing hepatomas after 11 months of exposure and Kanechlor 500 (similar in composition to Aroclor 1254) inducing hepatocellular carcinomas after 8 months of exposure. These lesions were shown to be reversible and specific for the dose (500 ppm) and chlorination of the PCB mixture.

In rats, Aroclor 1260 or its equivalent, Clophen A60, have produced hepatocellular carcinomas in three studies at doses of approximately 100 ppm, a dose which appears to represent the maximally tolerated dose for rats. A review of these three studies indicates that the tumors occur very late in the life of the animal, with a significant incidence of tumors only beginning to appear after about two years of exposure. Of interest is the fact that in all three studies the PCB treatment, while increasing the incidence of liver cancer, did not increase the total tumor incidence. The total tumor incidence was not increased in these studies because in each case the incidence of other tumor types had been significantly decreased. This suggestion of antitumor activity of PCBs has also been demonstrated in a study examining the effect of PCB exposure on the final tumor incidence in animals following the transplantation of the Walker 256 sarcoma. The effects of chronic PCB-treatment was not life-shortening, and in fact in two of the studies the morbidity and mortality of the animals was actually decreased by PCB treatment. Furthermore, while the tumors are described as malignant, i.e. hepatocellular carcinomas, in none of the three studies did the liver tumors metastasize to other organs even though metastases would be expected if the tumors were malignant. So from these studies it is evident that PCB-treatment does not increase the total cancer risk in these animals, rather it shifts the incidence of the type of tumors observed by significantly decreasing some tumor types while enhancing the incidence of liver tumors. Lastly, PCB mixtures of lesser chlorination, i.e. Aroclor 1254 and Clophen A30 (similar in composition to Aroclor 1242, see table 1.4, page 8, of Brinkman and DeKok, 1980), have been examined in two separate studies and found not to be carcinogenic. Thus, conclusions to be drawn from the rat data, like the mouse data, are specific for the dose and degree of chlorination of the PCB mixture being tested.

Possibly because PCBs produce liver hypertrophy, they enhance the tumorigenesis of certain liver carcinogens if given after the carcinogen in question has had an opportunity to initiate tumors. However, if the PCB exposure precedes or is concomitant with exposure to liver carcinogens the tumorigenic response is typically decreased, probably as a result of an enhanced metabolic detoxification of the carcinogen.

To summarize, the qualitative human relevance of the carcinogenic activity of PCBs based on the animal data is limited. The studies providing some evidence of its carcinogenic activity are specific for the degree of chlorination of the PCB mixture, the total tumor incidence is not increased, and the tumors produced occur only very late in the life of the animal and have no adverse effect on the morbidity or mortality of the animal. There are other considerations that limit concern for the carcinogenic data in rats as well. PCBs are not mutagenic, and the mechanism of tumorigenesis for these compounds therefore would appear to involve an epigenetic mechanism. There is also substantial evidence that the doses used to induce tumors in rats are hepatotoxic, and evidence indicating the neoplasms induced by PCBs are reversible if the exposure is terminated before the animal has been exposed for a considerable portion of the animal's lifespan. All of these findings seriously undermine the human relevance of the animal carcinogenicity data. This is particularly true as the human dosages from past and present human exposures are far lower than those used in the animal studies. Given these considerations, it is concluded that the animal data provides *sufficient evidence* of *limited human relevance* that PCBs of 60% chlorine content (e.g. Aroclor 1260/Clophen A60) are carcinogenic in animals. For PCBs having a chlorine composition of 54% (e.g. Aroclor 1254/Kanechlor 500) there is only *inadequate evidence* of carcinogenicity in animals because the larger and longer study in rats was negative, a finding suggesting that the reversible effects reported in mice may have resulted from a promotion of the substantial background incidence of liver tumors occurring in this species. For the remaining commercial PCB mixtures (i.e. Aroclor 1248, Aroclor 1242, Kanechlor 400, Kanechlor 300, and Clophen A30) there is either *no evidence* or *insufficient evidence* for the carcinogenicity of these mixtures in animals.

Evaluation of Epidemiological Evidence

Evidence for human carcinogenicity would be supplied if the two major epidemiological studies of PCB-exposed workers demonstrated a consistent increase in cancer mortality. This increase should show a positive correlation with exposure and evidence of latency. The evidence for carcinogenicity would be strengthened if the incidence of specific types or sites of neoplasms were consistently elevated. In reviewing the data from these two cohorts, none of these conditions are met.

While Bertazzi et al. (1986), in their study of Italian PCB-exposed workers, found a statistically significant elevation in the rate of cancer mortality among both male and female workers, the much larger study of Brown found no increase in cancer mortality. In the Brown study, there were higher-than-expected incidences of rectal and liver cancer. Evidence for an association between PCB exposure and these malignancy types cannot be considered strong, however, in that: 1) no cases of rectal cancer were observed after the initial report, suggesting that this increased rate was anomalous, 2) the number of cases of liver cancer observed in this study is not appreciably greater than expected when examined without total number of liver neoplasms used by Brown which includes those liver cancers that have metastasized from other organs, 3) in comparison, the study by Bertazzi et al. found only one case of liver cancer and no cases of rectal cancer in their cohort, and 4) the absence of a clear association with latency or relationship with duration of exposure.

The primary sites of neoplasms contributing to the higher-than-expected cancer mortality rates in the Bertazzi studies were located in the digestive system and the hematopoietic and lymphatic systems. These were not increased in the larger cohort reported by Brown. Further, in the Bertazzi cohort, there was no evidence of latency or relationship between cancer mortality and exposure to PCBs. It should also be noted that in the larger of the two subdivisions of their cohort, the female workers, differences in incidences of causes of death presumably unrelated to PCB exposure (viz., increases in accidental death and decreases in deaths from cardiovascular disease) were of similar magnitude as increases in death from malignant tumors. This suggests, at least for this group, that other confounding variables may exist.

In summary, epidemiological evidence for human carcinogenicity of PCBs is at present weak and mostly negative. As such, until larger epidemiological studies can be completed, the data must be considered *inadequate to characterize PCBs as human carcinogens*.

Summary Table

Aroclor 1260/Clophen A60 :

There is *sufficient evidence* of *limited human relevance* for the carcinogenicity of Aroclor 1260 in animals.

The human evidence for carcinogenicity of this compound is *inadequate* .

Aroclor 1254/Kanechlor 500 :

There is *inadequate evidence* for the carcinogenicity of Aroclor 1254 in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

Aroclor 1248/Kanechlor 400 :

There is *no evidence/insufficient evidence* for the carcinogenicity of Aroclor 1248 in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

Clophen A30/Kanechlor 300/Aroclor 1242 :

There is either *no evidence* or *insufficient evidence* for the carcinogenicity of these mixtures in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

BIOLOGICAL DATA RELEVANT TO THE EVALUATION OF CARCINOGENIC RISK TO HUMANS

1.0 Carcinogenicity and Related Studies In Animals

1.1 Studies in Mice

Nagasaki et al. (1972) initially examined the hepatocarcinogenic effects of Japanese brands of PCB fluids by feeding male mice dietary levels of 100 ppm, 250 ppm, and 500 ppm for 32 weeks. As this particular citation represents a short communication, detailed discussions of the experimental design and histopathological examinations of this investigation were lacking. An increased incidence of liver tumors was found in only one of the treatment groups, those mice receiving the 500 ppm diet of Kanechlor 500, where hepatomas were identified in 58% or 7/12 of the animals. The livers from animals in this group also contained nodular areas and many necrotic foci. In sharp contrast to these findings, no hepatomas and none of these histopathologic changes were observed in animals receiving lower doses of Kanechlor 500 or in any of the animals receiving Kanechlor 400 or Kanechlor 300.

The results of the above study also appear to have been reported in two other journals under Ito et al. (1973a&b). As in the previous study it was reported that male mice of the ddY strain were fed diets containing Kanechlor 300, Kanechlor 400 or Kanechlor 500 at dietary levels of 100, 250 or 500 ppm for 32 weeks. PCBs significantly increased the liver weights of the animals, and at the highest dose of Kanechlor 500 the liver:body weight ratio had increased 3-fold with histopathological examination of the livers revealing a focal hypertrophy in the centrilobular hepatocytes of non-neoplastic areas. Another change observed in all PCB treatment groups, except those receiving 500 ppm of either Kanechlor 500 or 400, was a marked amyloid degeneration of the liver in the space between the sinus endothelium and the hepatocytes. For some reason, the liver tumors in these more recent publications (Ito et al, 1973a&b) have been reclassified, and the tumors reported as hepatomas in the previous communication (Nagaski et al., 1972) are now listed as carcinomas (Ito et al., 1973 a&b). That is, in the group receiving 500 ppm of Kanechlor 500 (originally described as having hepatomas in 7/12 animals) nodular hyperplasia was found in 7/12 animals (58.3%) while 5/12 of the livers (41.7%) now have well-differentiated hepatocellular carcinomas. The carcinoma cells were reported to be comprised of irregularly shaped cells with pyknotic nuclei and occasionally had mitotic nuclei. The reason for changing the classification of these neoplasms from hepatoma to carcinoma is not provided. Of some interest to discussions of the carcinogenic potential of PCBs is the fact that all of the other doses of all three Kanechlors tested failed to produce even nodular hyperplasia. Thus, the tumorigenicity reported was quite specific for the dose and the extent of chlorination of the Kanechlor being tested.

Kimbrough and Linder (1974) examined the effects of PCBs in

ce and obtained results consistent with those found for the lesser-chlorinated Kanechlors of the preceding study. Groups of 50 male Balb/cJ mice were fed 300 ppm Aroclor 1254 for either 11 months or for 6 months followed by a 5 month recovery period. The authors pointed out that this dietary level was some 2,500 times the Food and Drug Administration's estimate of the average human daily PCB intake from food during the early 1970's. About one-half of the animals in each PCB treatment group died during the first 4 months, but this problem seems to have been unrelated to the PCB treatment as control animals suffered a similar incidence of mortality. Of the 22 mice surviving the 11 month feeding study, all animals had hepatomegaly with the liver representing approximately 25% of their body weight in comparison to the 5.8% liver:body weight ratio measured for the control animals. Adenofibrosis (i.e. cholangiofibrosis) was observed in all 22 livers taken from mice fed PCBs for 11 months, however 13/68 livers taken from control animals had occasional, small areas of necrosis and fibrosis. Of the livers taken from animals receiving 300 ppm Aroclor 1254 for 11 months, the nuclei were enlarged, hyperchromatic and atypical. The cytoplasm was either smooth or vacuolated, and the Kupffer cells contained a brown pigment. Some of these livers had extensive necrosis and fibrosis. In each of the livers, several areas of hepatocytes had been replaced by proliferating epithelial cells which formed ducts and often produced mucus. In addition to these histological changes, 10 hepatomas were found in 9 of these livers. The tumors were described as well circumscribed and surrounded by compressed parenchyma or strands of fibrosis. In the 24 surviving mice fed PCBs for only 6 months followed by a 5 month recovery period, only one liver contained a hepatoma. Yet, liver fibrosis was observed in two-thirds of these animals and hepatocellular necrosis was evident in most of the livers.

1.2 Studies in Rats

Kimura and Baba (1973) exposed 10 male and 10 female Donryu rats to a variable dietary level of Kanechlor 400 for 400 days. The diet initially contained 38.5 ppm and was fed to the animals for 4 weeks; the dietary level was then doubled and provided for the following 8 weeks; the initial dietary level was then increased 4-fold and provided for 3 weeks; it was then increased 8-fold and fed to the animals for another 3 weeks; finally it was increased to 16 times the initial level and fed to the animals for 3 more weeks. This last increase in the dietary levels of PCBs (a level that was approximately 616 ppm) was found to be too toxic and caused a considerable weight loss in the animals. In response to the toxicity observed at this dose, the dietary level was reduced to 462 ppm for the remaining 32 weeks of the study. Further complicating interpretations of this study is the fact that animals died or were sacrificed at various times throughout the experiment, therefore the total PCB dose each animal received may differ. In general, the total amount ingested was thought to be 1300-1800 mg for the group of male animals and 1100-1500 mg for the female animals. Microscopically the livers of all of the treated animals

contained fatty degenerative changes, and while 6/10 of the livers from female animals had adenomatous nodules, none of the livers of the male animals contained such nodules. However,the liver nodules observed in the female animals do not appear to be related to the PCB treatment as 2/5 (40%) of the livers from the control female animals also contained adenomatous nodules.

In a second study Kimura et al. (1976) fed 12 female Donryu rats diets containing 400 ppm Kanechlor 400 for 6 months. The estimated dose corresponded to a total of 531 mg of PCBs during this period. Eight of the 12 animals were then sacrificed 590 days after the feeding began. None of these animals developed hepatocellular carcinoma, and 9/12 of the livers were normal in appearance, suggesting that the degenerative changes observed in the previous study are reversible.

Ito et al. (1974) fed male Wistar rats Kanechlor 300, Kanechlor 400 or Kanechlor 500 at dietary levels of 100, 500 or 1,000 ppm for up to 52 weeks. No hepatocellular carcinoma was found in the livers of any of the treated rats. The highest dose of all three Kanechlors did produce a cholangiofibrosis of the liver, but this effect was not observed at the lower doses with any of the Kanechlors. Nodular hyperplasia was observed in 30-40% of the rats exposed to the two highest doses of Kanechlor 500 (i.e. doses of 500 ppm and 1.000 ppm) and in animals receiving a diet containing 1,000 ppm of Kanechlor 400. Oval cell proliferation and proliferation of the bile duct cells were observed in all treatment groups. Hypertrophy of the centrilobular cells was also evident in animals receiving the highest dose of the two most heavily chlorinated PCB mixtures. Fatty changes and fibrosis were also observed in the livers of animals of several of the treatment groups. The fatty changes, hypertrophy and fibrosis of the liver all tended to be present and correlate with the observation of nodular hyperplasia, suggesting that these changes may have been contributory factors.

Kimbrough et al. (1975) published the first major positive study demonstrating that Aroclor 1260 can produce hepatocellular carcinoma in the rat. In this study 200 female Sherman strain rats were fed Aroclor 1260 at a dietary level of 100 ppm for approximately 21 months. There was a statistically significant 6-7% decline in the weight gain of the animals exposed to PCBs in this study suggesting that the dose used approximated the maximally-tolerated dose. The incidence of the histopathological findings from this study are summarized in Table 1. The most consistent histopathologic difference in the PCB treatment group was the finding of hyperplastic or neoplastic nodules in 144/184 (78%) of the livers. More importantly, however, was the finding of hepatocellular carcinoma in 26/184 (14%) of the PCB-treated animals. The tumors were well-differentiated neoplasms of the trabecular type, except in three of the animals which had tumors with a glandular, papillary pattern. Foci of coagulative necrosis were occasionally observed in the cancerous areas, but there was no fibrosis or other evidence of chronic degenerative changes. Tumors

in areas other than the liver were not listed as significantly different, however, in some cases there was a substantial decrease in the tumor incidence of other tissues, e.g.

TABLE 1

Results of Carcinogenesis Bioassay of Aroclor 1260

		Tumor Incidence	
Organ/Tissue	Tumor Type	Control Animals	Aroclor 1260 Animals
Liver	Hepatocellular carcinoma	1/173	26/184
	Neoplastic nodules	0/173	144/184
	Areas of cytoplasmic alteration	28/173	182/184
Thyroid gland	Parafollicular cell tumor	37/160	18/166
Pituitary gland	Chromophobe adenoma	41/153	28/139
Mammary gland	Fibroadenoma	17/173	13/184
Ovary	Granulosa theca cell tumor	5/149	0/163

Adapted, in part, from Kimbrough et al. (1975)

as in the case of parafollicular cell tumors of the thyroid. The total incidence of extra-hepatic tumors in the Kimbrough et al. (1975) study was 134/173 (74%) in control animals versus 110/183 (60%) in Aroclor 1260 treated animals. So while PCBs had significantly increased the liver carcinoma incidence, the total tumor incidence (78% in control animals -vs- 74% in Aroclor animals) in PCB-treated animals was actually slightly less than the incidence for control animals. The PCB treatment was also not life-shortening; on the contrary, about twice as many control animals had died for various reasons before the experiment was terminated at 23 months.

Calandra (1976) at the 1975 PCB conference sponsored by the EPA reported the findings of several chronic studies performed by a commercial laboratory for Monsanto. While these studies have never been published the results were recently reviewed by Levinskas (1981). In this study 1,000 rats were divided into ten groups of 100 animals (50 of each sex) and nine of these groups were exposed to Aroclors 1242, 1254 or 1260 at dietary levels of 1, 10 or 100 ppm. Apparently five animals of each sex were sacrificed at 3, 6 and 12 months with about 35 animals to be killed at the end of the two year study (EPA, 1980). In the animals sacrificed early, only

one nodular hyperplasia was observed and it was in the group fed 100 ppm of Aroclor 1260 for 12 months. Mortality in this study was high and approximately one-third of the 105 animals anticipated to be exposed for two years at each dietary level died. Hepatomas were observed in 7/25 livers from animals fed 100 ppm Aroclor 1260, in 4/26 fed Aroclor 1254, 3/19 fed Aroclor 1242, and only 1/168 animals receiving the 1-10 ppm diets. Nodular hyperplasia was twice as prevalent as hepatomas in the high dose animals, particularly in the Aroclor 1254 group.

In 1978 the National Cancer Institute examined the carcinogenic potential of Aroclor 1254 (NCI, 1978). Groups of 24 male and 24 female Wistar rats were fed Aroclor 1254 at dietary levels of 25, 50 or 100 ppm for 105 weeks. Clinical signs of toxicity including hair loss, facial edema and cyanosis occurred by week 72 in the high dose animals and the mean body weights were roughly only 2/3-3/4 that of their respective controls. This decrease in body weight exceeds the no more than 10% weight loss guideline for the estimated maximally tolerated dose that is part of the NCI guideline for cancer bioassays (NCI, 1979). Several histopathologic changes occurred in the livers of animals receiving PCBs that appeared to be related to the PCB treatment, particularly the incidence of hyperplastic nodules and adenomas. Male animals had one hepatocellular carcinoma in the 50 ppm group and only 2 in the 100 ppm group. Although the incidence of these tumors was not significant, the occurrence of proliferative lesions did appear to be dose related. In reviewing this bioassay, the Data Evaluation/ Risk Assessment subgroup of the Clearinghouse on Environmental Carcinogens responsible for providing peer review of NCI studies concluded the following :

> "It is concluded that, under the conditions of the bioassay, Aroclor 1254 was not carcinogenic in Fischer 344 rats; however, a high incidence of hepatocellular proliferative lesions in both male and female rats were related to treatment. In addition, the carcinomas of the gastrointestinal tract may be associated with treatment in both males and females. Based on the liver proliferative lesions in the treated rats and published reports, it is suggested that Aroclor 1254 may be a tissue promoter."

Morgan et al. (1981) have taken the same tissue sections that originated in the NCI bioassay (NCI, 1978), stained the stomach sections for alkaline phosphatase, and then re-sectioned these tissues for histological evaluation in conjunction with those provided in the NCI study itself. The final incidence of alkaline phosphatase rich areas was 6.4% in controls, 10.4% in animals fed 25 ppm Aroclor 1254, 16.7% in the 50 ppm group, and 35.4% in the 100 ppm group. These changes were most often noted in the pyloric region of the stomach and duodenum (88% of the lesions were found in these areas), suggesting a toxicity specific to the cells of these areas. Gastric adenocarcinomas comprised six of the 33 total

lesions identified in these slices. Three were found in tissues from the 50 ppm treatment group and two in the animal group fed 100 ppm. Thus, the increased incidence of this tumor was not related to the dose of PCBs. The remaining 27/33 lesions identified in this study were described as intestinal metaplasia. The authors concluded that the actual number of lesions they believed should have been observed in the G.I. tract tissue sections of the NCI study was twice the number of lesions reported in the original NCI study. Further, and on the basis of their findings, the authors of this paper concluded that chronic oral Aroclor 1254 exposure may lead to the induction of intestinal metaplasia, and possibly to adenocarcinoma of the glandular stomach of the Fischer 344 rat (Morgan et al., 1981).

Ward (1985) has also published a review of the slides originating from the NCI bioassay. In addition to the aforementioned dose-related depression of body weight, Ward (1985) also discusses, in some detail, the substantial decrease in animal survival that occurred in this study. While the survival rate in control animals and in the treatment group receiving 25 ppm was 92% and 83%, respectively, only 58% of the animals receiving the 50 ppm diet and 46% of the animals fed diets containing 100 ppm survived to the end of the bioassay. Focal hyperplasia was of the eosinophilic type and was only observed in PCB-treated animals. If compression was found on two sides of the neoplasm, Ward diagnosed the lesion as hepatocellular adenoma. A total of 13 eosinophilic, basophilic or vacuolated adenomas were identified. All of these occurred, with one exception, in those animals fed the two higher dietary levels of Aroclor 1254 and the occurrence of adenomas was slightly greater in the male animals (8/13). Similar to the findings of the NCI (1979) report, only two liver carcinomas were identified, both occurred in male animals receiving the 100 ppm diet. Ward (1985) also reported that Aroclor 1254 increased the incidence of intestinal metaplasia and gastric adenocarcinoma. As in his earlier report with Morgan and Hartman (Morgan et al.,1981), the change in adenocarcinoma was neither significant nor dose-related. A significant increase in intestinal metaplasia was only observed in the 100 ppm dose group. Thus, the Ward (1985) study is still consistent with the previous NCI (1979) bioassay. No statistically significant increase in liver cancer or cancer of other tissues was observed, but the 100 ppm dose does lead to significant intestinal metaplasia. Ward mentions the fact that the liver lesions he observed were predominantly of the eosinophilic type rather than the basophilic type generally observed in the control animals. Based on these changes Ward proposes the idea that these data may suggest that PCBs are capable of initiating liver tumors rather than promoting the background tumor incidence. Yet, in contradiction of his suggestion, Ward also makes note of the fact that inducing agents like PCBs and phenobarbital cause a proliferation of the smooth endoplasmic reticulum (SER) of the liver. As a proliferation of the SER gives rise to an eosinophilic appearance of the cytoplasm, the liver hypertrophy and induction of microsomal enzymes associated with PCB exposure provides an obvious explanation for the basophilic to eosinophilic change Ward noted in

the cellular appearance of the liver tumors of PCB-treated animals.

Schaeffer et al. (1984) used a total of 432 weanling Wistar rats to examine the effects produced by chronically feeding rats Clophen A60 (equivalent to Aroclor 1260) or Clophen A30 (similar in composition to Aroclor 1242; Brinkman and DeKok, 1980). The study consisted of three groups. Group 1, a control group of 139 animals receiving the normal diet, Group 2 with 152 animals receiving a diet containing 100 ppm of Clophen A30, and Group 3 which consisted of 141 animals fed a diet containing 100 ppm of Clophen A60. After day 801 animals were randomly selected and killed, and the experiment was terminated on day 832. The Clophens used in this study were reported to be free of any chlorinated dibenzofuran contamination, but the level of detection for this analysis was not specified. In those animals necropsied prior to day 800, hepatocellular carcinomas were only identified in the PCB treatment groups, one in Group 2 and a total of 9 were observed in Group 3. This latter number was statistically significant for the Clophen A60 treatment, but represented a liver cancer incidence of only 7% for the entire group. In contrast, the incidence of thymoma was significantly reduced by PCB treatment declining from 12% in the control group to 3-4% in the treatment groups. Likewise the total number of the remaining types of neoplasms was significantly reduced by the PCB treatment, with Clophen A60 causing the greatest reduction (from 52 in controls down to 18 in the Clophen A60 group). The final results of this study are shown below in Table 2. The incidence of hepatocellular carcinoma was significantly increased only in those animals receiving the Clophen A60. Thus, the results of this study were consistent with the previous rat studies, i.e. Aroclor 1260, or its equivalent, was reported to have induced hepatocellular carcinoma in rats while a lesser chlorinated PCB mixture was not carcinogenic.

Table 2

Frequency of Hepatocellular Alterations Induced by Chronic Feeding Studies with Clophen A30 and Clophen A60

	# of Foci	Neoplastic Nodules	Hepatocellular carcinoma
Controls (group 1)	6/131 (4.5%)	5/131 (3.8%)	1/131 (0.8%)
Clophen A30 (group 2)	63/130* (48%)	38/130* (29%)	4/130 (3%)
Clophen A60 (group 3)	3/126 (2.4%)	63/126* (50%)	61/126* (48%)

* denotes a significant difference from the control value ($P<0.05$)

In a letter to the editor, Young (1985) comments on several additional, interesting and important aspects of the Schaeffer et al. (1984) study that were not noted by the authors of this paper. Young's analysis of this data focused on the effects of PCB exposure on tumor incidence in liver, on tumor incidence in extra-hepatic tissues, and on mortality. The tables generated by Young (1985) are provided below in Table 3.

Table 3

The Incidence of Hepatocellular Carcinoma, Other Neoplastic Lesions, and Mortality by Time Period

Time Interval (days)	Treatment Group: Control	Clophen A30	Clophen A60
A. Hepatocellular carcinoma			
301-400	0/137	0/150	0/135
401-700	0/111	0/122	0/115
701-800	0/92	1/107	9/85*
B. Other Neoplasms			
301-400	0/137	1/150	2/150
401-700	32/111	11/122*	9/115*
701-800	30/92	15/107*	7/85*
C. Incidence and Percentage(%) of Mortality			
101-400	2/139 (1.4%)	2/152 (1.3%)	3/141(2.1%)
401-700	45/137(32.8%)	43/150(28.7%)	23/138(16.7%)
701-800	39/92 (42.4%)	20/107(18.7%)	30/85 (35.3%)
1-800	86/139(61.9%)	65/152(42.8%)*	56/141(39.7%)*

* Different from control value ($P<0.05$), χ^2

Young points out that the Schaeffer et al. (1984) study actually demonstrated three things. These are : 1) that a significant increase in hepatocellular carcinoma occurred only in the animals receiving Clophen A60, 2) that PCB treatment resulted in a significant decrease in other neoplastic lesions, and 3) that PCB treatment significantly increased the chances of survival of the animals. Given these findings, Young states that it is difficult to conclude, given the balance of the data, whether or not the PCB treatment was in fact detrimental to the rats in light of the fact that the PCB treatment significantly enhanced the rate of survival

and significantly decreased the total tumor load of the exposed rats. In essence, he questions the human relevance of tumors which occur only very late in the life of the animal, are not life-shortening and do not metastasize to other organs of the body. Thus, to quote Young :

> "If the purpose of long-term studies is to extrapolate to humans, then one finds it difficult to infer dire consequences to humans when the treatment is beneficial in the model system. Is the model only useful for inferring bad events? The model should be equally valid for detrimental and beneficial effects."

Lastly, the analysis of Young also calls to question a suggestion made by Schaeffer et al. (1984), which was that the decrease in the incidence of thymoma might be caused by immunosuppressive effects of PCBs. While PCBs cause thymic atrophy at certain doses, any proposed immunosuppressive effect cannot be considered to have a significant clinical impact when the treated animals did not ultimately suffer a greater incidence of morbidity or mortality from either infectious diseases or the cancer induced by this treatment.

The last major rat study reported is that of Norback and Weltman (1985). These investigators fed 70 male and 70 female Sprague-Dawley rats a diet containing 100 ppm Aroclor 1260 for 16 months followed by a reduction to 50 ppm for the next 8 months. The animals were then fed a control diet for the remaining 5 months of their lives. All results were compared to a control group which initially consisted of 126 animals, 63 of either sex. At various time points throughout this study two control animals of each sex and three PCB-treated animals of each sex (10 animals in total) were anesthetized with ether and the medial left lobe of the liver of each animal was surgically removed. These tissue samples were taken at 1, 3, 6, 9, 12, 15, and 18 months. At 24 months a similar group was killed and at the end of 29 months all remaining animals were sacrificed. The induction of liver hypertrophy in the centrilobular area of the lobule was evident at the first observation period made one month after the PCB diet was initiated. By the 18th month the liver:body weight ratio had increased from 4% to 12% in the female animals. Macroscopically these investigators noted evidence characteristic of neoplastic nodules near the capsular surface, hepatocellular carcinomas and adenofibrosis. In the PCB-exposed group, the observed lesions appeared in the following sequence : centrilobular hypertrophy at 1 month, foci of cells appeared at 3 months, foci of altered cells in the centrilobular and midzonal regions at 6-9 months, neoplastic nodules appeared at 12 months, trabecular carcinoma was observed after 15 months and adenocarcinoma at 24 months. Simple cystic cholangioma and adenofibrosis appeared in animals 18-23 months after the exposure began. There was no evidence of metastases to the lungs. All trabecular carcinomas had cell arrangements with a glandular, ductal or cystic pattern and all adenocarcinomas had some elements of the trabecular pattern of growth. The lumens of

the adenocarcinomas were the apparent result of cellular necrosis.

The incidence of tumors in animals 18 months or older are presented below in Table 4. It should be noted that 7-8 animals

Table 4

Incidence of Hepatocellular Neoplasms

	Incidence or % Tumors Observed (The actual number of animals with tumors)		
	Male	Female	Total
A. Control Animals	(N=32)	(N=49)	(N=81)
Trabecular carcinoma	0%(0/32)	0%(0/49)	0%(0/81)
Adenocarcinoma	0%(0/32)	0%(0/49)	0%(0/81)
Number negative	100%(32/32)	98%(48/49)	99%(80/81)
B. Aroclor 1260 Animals	(N=46)†	(N=47)††	(N=93)
Trabecular carcinoma	4%(2/46)	40%(19/47)	23%(21/93)
Adenocarcinoma*	0%(0/46)	51%(24/47)	26%(24/93)
Neoplastic nodule only	11%(5/46)	4%(2/47)	8%(7/93)
Number negative	85%(39/46)	4%(2/47)	44%(41/93)

† The total number includes 8 animals that had received a partial hepatectomy during the first 18 months.
†† The total number includes 7 animals that had received a partial hepatectomy during the first 18 months.
* Animals with both trabecular carcinoma and adenocarcinoma were placed only in the adenocarcinoma group.

sacrificed after 18 months, i.e. at least 15% of the group of animals in which late developing tumors were observed, had received a partial hepatectomy during the first 18 months. The effect of this cannot be determined from this experiment, but partial hepatectomy has been used as a promotional stimulus to increase the incidence of liver tumors induced by other carcinogens. Therefore, it is unfortunate that the authors did not note or describe the possible influence that this might have had on the final tumor incidence measured. Another important factor to consider, and one which is not readily apparent from Table 4, is that almost all of the tumors reported in this study were very late-developing tumors. In Table 1 of their paper, only 4 trabecular carcinomas and only 1 adenocarcinomas had developed between the 18- and 24-month sacrifices. Thus, 35/41 or some 85% of the liver tumors observed in this study developed in the last 25-29 month period of the study. These tumors had not metastasized to other organs, and none appear to have been life-shortening. Concerning this last aspect, unfortunately no information is given concerning the cause of death for any animals dying early, or concerning the number of animals lost. But from the data supplied

t would appear that a number of early deaths occurred only in the group of male control animals. These same observations were also noted by the authors, who stated in the discussion of this paper :

> " Although the tumors met the morphological criteria for malignancy, their biologic behavior was relatively unaggressive. The neoplasms did not metastasize to distant organs nor invade blood vessels. Mortality of the animals was not increased. The lack of greater morbidity or mortality is likely due to slow progression of the neoplastic process and late appearance and slow growth of the hepatocellular carcinoma."

The authors further noted that it remains to be established whether PCBs have an initiating effect or whether the neoplasms observed result from the promotion of a background incidence of initiated cells.

1.3 Studies In Other Species

Calandra (1976) reported, in summary form, a carcinogenicity bioassay performed in dogs. Groups containing 4 male and 4 female dogs were fed 1, 10 or 100 ppm of Aroclors 1242, 1254 or 1260 for two years. While this exposure interval is for a period of time that is considerably less than the lifetime of the species being tested, no remarkable findings were reported.

1.4 The Effects of PCBs on Other Liver Carcinogens

PCBs have been tested in numerous studies to determine whether they alter the carcinogenicity of other chemicals. The evidence obtained from such studies is sometimes inconclusive, and in several instances similar studies have produced conflicting results. The results of have been summarized in the following paragraphs and in table 5 located on page 13 :

- A number of studies indicate that PCBs administered after a carcinogenic dose of a liver carcinogen enhance the incidence of tumors. Nishizumi (1976) found that Kanechlor 500 promotes liver tumors initiated by diethylnitrosamine. Kimura et al. (1976) demonstrated that Kanechlor 400 can promote the liver cancer induced by 3'-methyl-4-dimethylaminoazobenzene (MeDAB). Ito et al. (1978) initiated liver tumors with N-2-fluorenylacetamide and then increased tumor incidence by feeding the rats diets containing 1,000 ppm of PCBs. Preston et al. (1981) have reported that Aroclor 1254 with or without polychlorodibenzofuran contaminants can promote the liver cancer initiated by diethylnitrosamine. Osterle and Deml (1984) demonstrated that this effect can be produced in weanling rats, although the immature animal is less sensitive. Deml et al. (1983) have also found that an initial PCB pretreatment to increase

oxidative metabolism, followed later by PCB administration for its promotional effects, increased the number of rat liver preneoplastic islands. Similarly, Pereira et al. (1982) demonstrated that Aroclor 1254 promoted the number of diethylnitrosamine-induced enzyme-altered foci in rat livers. Recent work by Deml and Oesterle (1987) has shown that while PCBs are capable of promoting the liver tumors initiated by diethylnitrosamine, the threshold for this effect is about 1 mg/kg/day and correlates strongly to the induction of MFO activity and subsequent liver hypertrophy that the induction of drug metabolism normally produces (Greim et al., 1985).

- In contrast to the above studies, Gans and Pitauro (1986) reported Aroclor 1254 increased the incidence of liver scarring in mice fed diethylnitrosamine, but PCB treatment did not enhance the formation of liver nodules. Arai et al. (1983) also reported that PCBs administered to rats following an initial exposure to dimethylnitrosamine increased the incidence of liver tumors, however, this same dosage regimen decreased the incidence of kidney tumors produced by DEN.

- Conflicting results have been reported in trout where the effect of PCBs is dependent upon the time of PCB administration as well as the carcinogen it is administered with. For example, Hendricks et al. (1980) found that PCBs did not promote the incidence of liver tumors induced by aflatoxin B_1 in trout, even though the trout were fed a diet containing PCBs for 12 months after the aflatoxin B_1 exposure. However, other studies by Shelton et al. (1984a) have demonstrated that PCBs exert an inhibitory effect on the incidence of liver tumors initiated by aflatoxin B1 when PCBs are co-administered with this liver carcinogen. Alternatively, trout fed diets containing 1100 ppm diethylnitrosamine + 100 ppm of either Aroclor 1242 or Aroclor 1254 had a greater incidence of liver cancer than did trout fed diets that only contained 1100 ppm DEN (Shelton et al., 1984b).

- Loury and Byard (1983) observed that Aroclor 1254 enhanced the DNA repair response of amino acid pyrolysate mutagens in primary hepatocyte culture when the PCBs were given three days prior to cell isolation. Thus, it appears that these effects are related to a PCB enhancement of oxidative metabolism. In contrast, Nesnow et al. (1981) found that Aroclor 1254 did not enhance the benzo(a)pyrene-mediated transformation of C3H10T1/2CL8 mouse embryo fibroblasts even though a number of other chemicals known to induce drug metabolism were positive in this test.

- PCB exposure <u>in utero</u> and during nursing protected

Wistar rats and CD-1 mice from diethylnitrosamine-induced liver tumors (Nishizumi, 1980, Anderson et al., 1983). Likewise Kimura et al. (1976) found that PCBs given before or during MeDAB exposure decreased the incidence of MeDAB-induced liver cancer. Similarly, Makiura et al. (1974) have shown that the co-administration of PCBs with the liver carcinogens MeDAB, N-2-fluorenylacetamide and diethylnitrosamine decreases tumor incidence. Trout fed diets containing PCBs prior to their exposure to aflatoxin had a reduced frequency of liver tumors induced by the aflatoxin (Hendricks et al. 1977). This effect is perhaps explained by Stott and Sinnhuber (1978) who demonstrated that PCBs reduced the bioactivating ability of trout microsomal fractions used to test the mutagenicity of aflatoxin B_1.

- In dermal initiation-promotion assays PCBs have been found to diminish the initiating activity of certain skin carcinogens and to be devoid of any promoting activity (DiGiovanni et al., 1977; Berry et al., 1978). Similarly Hayes et al. (1985) examined the effects of PCBs on the proliferating hepatocytes in livers of young animals or adult animals after partial hepatectomy. They concluded that short-term PCB exposures do not have an initiating action in an *in vivo* assay that detects both hepatic and extra-hepatic initiating carcinogens.

- In contrast to its reportedly adverse effects on the immune system, Kerklivet and Kimeldorf (1977a&b) demonstrated that PCBs administered either in the diet or by injection reduced the size of Walker 256 carcinosarcomas transplanted to rats and increased the lifespan of the PCB-treated animals.

Table 5

Summary of PCB Interactions With Other Carcinogens

Study	Carcinogen	Results
A. PCBs Administered After the Carcinogen :		
Nishizumi (1976)	diethylnitrosamine	promotion (liver tumors)
Kimura et al., (1976)	Me-DAB	promotion (liver tumors)
...to et al., (1978)	N-2-fluorenylacetamide	promotion (liver tumors)

Hendricks et al., (1980)	aflatoxin B1	no effect
Preston et al., (1981)	diethylnitrosamine	promotion (liver tumors)
Pereira et al., (1982)	diethylnitrosamine	promotion (liver tumors)
Deml et al., (1983)	benzo(a)pyrene	promotion (liver tumors)
Arai et al., (1983)	dimethylnitrosamine	promotion (liver tumors) inhibition (kidney tumors)
Gans & Pitauro, (1986)	diethylnitrosamine	liver scarring no effect (liver nodules)
Deml & Oesterle, (1987)	diethylnitrosamine	promotion (liver tumors) threshold dose (1 mg/kg/day)

B. PCBs Administered Before or With the Carcinogen :

Makiura et al., (1974)	Me-DAB	less liver tumors
	N-2-fluorenylacetamide	less liver tumors
	diethylnitrosamine	less liver tumors
Kimura et al., (1976)	Me-DAB	less liver tumors
Hendricks et al., (1977)	aflatoxin B1	less liver tumors
Nishizumi (1980)	diethylnitrosamine	less liver tumors
Anderson et al., (1983	diethylnitrosamine	less liver and lung tumors
Shelton et al., (1984a)	aflatoxin B1	less liver tumors
Shelton et al., (1984b)	dimethylnitrosamine	promotion

Me-DAB = 3'-methyl-4-dimethylaminoazobenzene

It is concluded from these studies that the liver hypertrophy induced by PCBs can and does enhance the carcinogenic response of initiating hepatocarcinogens if the PCBs are given some time after the liver cells have been initiated by the other chemical.

wever, this effect has also been demonstrated for phenobarbital and is no doubt also true for most other liver enzyme inducing chemicals that produce a hypertrophy or hyperplasia of the liver. Additionally, recent work by Deml and Oesterle (1987) has shown that the promotional effects of PCBs has a threshold, as one might expect, and this threshold appears to be well above the dose commonly encountered by most persons. In contrast to its promotional effects on liver carcinogens known to be initiators, PCBs also act as anticarcinogens when administered prior to the exposure of an initiating carcinogen. This effect is most probably the result of PCB induction of liver metabolism which then acts to decrease the carcinogenic effects of a number of chemicals by increasing their degradation and elimination.

2.0 Epidemiological Studies of PCB Exposure and Cancer Mortality

Published epidemiological studies addressing the carcinogenic potential of PCBs have been confined to two cohorts, both consisting of employees of plants where PCBs were used in the manufacture of capacitors.[Note : Bahn et al. (1976), in a letter to the editor, reported an analysis of eight deaths among 92 research and development workers at a refinery, and suggested an association between PCB exposure and incidence of malignant lanoma. The extremely limited size of their cohort, the inability adequately assess exposure of the cohort to other, potentially carcinogenic substances or other known risk factors, and the inconsistency of their observations with the results from the two larger cohorts, precludes any utility of this report in the assessment of the human carcinogenicity of PCBs.]

Brown and Jones (1981) have published a retrospective cohort mortality study of employees from two plants. Minimal exposure period to PCBs was three months, and the types of PCBs used were Aroclors 1016, 1242, and 1254. Workers who had potential exposure to trichloroethylene, which was also used in both plants, were excluded from the cohort. Of the 2567 employee cohort, 163 were known to be deceased and the vital status of 55 (2%) was unknown.

Higher-than-expected incidences of cancer in the rectum (primarily among females in Plant #2) and liver were noted, although the differences were not statistically significant. The overall incidence of malignancies observed was slightly less than expected. None of the causes of death showed a clear association with latency, and no relationship was observed between the duration of employment in jobs involving PCB exposure and risk of mortality due to cancer. The causes of mortality and incidences of specific malignant neoplasms are summarized in Table 6.

The study of mortality in this cohort has recently been updated, with the number of deceased increasing from 163 to 295

(Brown, unpublished data). There were no additional deaths from cancer of the rectum which brought the observed incidence of this form of neoplasm closer to the expected value. The number of observed malignancies of the liver was significantly greater than expected. However, a number of these malignant neoplasms (3/5) appear to have been the result of metastasis from other sites. Thus, like the rectal cancer the standard mortality rate (SMR) for liver cancer originating in the liver has been substantially lowered by this larger cohort. A higher-than-expected incidence of neoplasms was not noted for any other tissue. The overall incidence of malignancies observed was less than expected. There was no apparent relationship between duration of employment in "PCB-exposed" jobs or total employment and cancer mortality of any type.

Table 6

Major Causes of Death for Plant Workers Exposed to Aroclor 1016, 1242, and 1254

Causes of Death	Observed/Expected	Standard Mortality Rate
	All Causes of Mortality	
All malignant neoplasms	39/43.79	(89)
Diseases of nervous system	11/12.55	(88)
Diseases of circulatory system	60/62.93	(95)
Accidents	13/18.29	(71)
All other causes	40/44.79	(80)
Total mortality for all causes	163/182.35	(89)
	Malignant Neoplasms	
Cancer of stomach	1/1.66	(60)
Intestine	4/4.03	(99)
Rectum	4/1.19	(336)
Liver	3/1.07	(280)
Pancreas	1/1.90	(53)
Respiratory	7/7.98	(88)
Breast	7/6.84	(102)
Lymphatic	2/4.34	(46)
Other	10/14.78	(68)

Source: Brown and Jones (1981)

The first study of the second major cohort was reported by Bertazzi et al. in 1981. Inclusion in the cohort was limited to workers in the production department who had been employed for at least six months. There were a total of 1310 workers included in

the study, and exposure was primarily to Aroclor 1254 and Pyralene 476. While some members of the cohort were also exposed to trichloroethylene, alkylbenzene, and epoxy resins, the number was stated to be few. A total of 27 deaths were observed in the cohort between 1954 and 1978. The overall incidence of cancer observed was significantly higher than expected (14 versus 5.65 expected), contributed primarily by elevated (though not statistically significant) incidences of neoplasms of the digestive organs and lymphatic and hematopoietic systems.

Bertazzi et al. (1986) have recently published an update of their study of PCB-exposed capacitor workers with a modified cohort. Non-production workers at the plant were added to the cohort, and the minimum period of employment was reduced from six months to one week. The size of the cohort was consequently increased to 2100, and the number of deceased to 64. The results from male and female workers were analyzed separately, and are summarized below in Tables 7 and 8.

Table 7

Mortality From Selected Causes of Male Workers Exposed to PCB

		Reference Cohort			
		National Mortality Estimates		Local Mortality Estimates	
Cause of death	Observed	Expected	SMR	Expected	SMR
All causes	30	27.8	108	29.8	101
Malignant tumors	14	5.5	253[a]	7.6	183[b]
Cancer of G.I. tract	6	1.7	346[c]	2.2	274[d]
Lung cancer	3	1.2	250	1.6	187
Hematologic neoplasms	3	0.8	375	1.1	263
Cardiovascular disease	8	7.9	101	9.4	95
Accidents	6	6.8	88	5.8	103

Confidence limits (95%) [a]= 144-415 [c]= 141-721
[b]= 104-300 [d]= 112-572

For males, mortality from cancer was significantly greater than expected (14 versus 5.5 expected). Among specific tissues, the observed incidence of neoplasms of the GI tract was significantly greater than expected, and hematologic neoplasms were also greater

than expected but not statistically significant. Overall mortality was only slightly greater than expected. Among females, higher-than-expected incidences of overall mortality, mortality from cancer, and incidence of hematologic neoplasms were observed. These differences were of statistical significance when the cohort was compared to local mortality rates but not national mortality rates. While neoplasms were grouped by organ-system for statistical analysis, specific tumor sites were noted. Among both male and female members of the cohort, there was one case of liver cancer and no cases of rectal cancer. There was no apparent association between duration of exposure, latency, and year of first exposure for any of the causes of mortality.

Table 8

Mortality From Selected Causes of Female Workers Exposed to PCB

		Reference Cohort			
		National Mortality Estimates		Local Mortality Estimates	
Cause of death	Observed	Expected	SMR	Expected	SMR
All causes	34	25.8	132	16.5	206[a]
Malignant tumors	12	7.7	156	5.3	226[b]
Hematologic neoplasms	4	1.5	266	1.1	377[c]
Cardiovascular disease	2	4.7	42	3.0	66
Accidents	9	4.0	225	4.0	225

Confidence limits (95%) [a]= 145-285 [c]= 115-877
[b]= 123-385

From Bertazzi et al., 1986

3.0 Other Relevant Human Data

3.1 Health Effects Information from Accidental Ingestion

In 1968, the ingestion of PCB-contaminated rice oil in Japan resulted in an outbreak of chloracne and other symptoms which were later termed "Yusho". A similar large-scale incident of ingestion of contaminated rice oil occurred approximately ten years later in Taiwan with similar symptoms. The most common symptoms observed in rice oil poisoning victims included increased discharge from the eyes, swelling of the eyelids, accompanying visual disturbances, fatigue and malaise, headache, and symptoms suggestive of peripheral neuropathies (numbness of the limbs and pruritis) (Higuchi, 1976; Okumra, 1984; Lu and Wong, 1984). Other notable

symptoms reported included acneform lesions and hyperpigmentation. The incidence of hyperpigmentation was high (9/10) among offspring born to mothers pregnant during ingestion of the contaminated rice oil (Higuchi, 1976). This hyperpigmentation gradually faded over the 2-5 months after birth. Developmental tooth and bone defects were noted among some of the offspring (e.g. eruption of teeth at birth, larger-than-usual frontal and occipital fontanelles, and maintenance of a wider sagittal suture than usual). However, postnatal physical and mental development of infants born to contaminated rice oil-exposed mothers paralleled that of healthy infants (Higuchi, 1976).

While it was originally assumed that the rice oil was contaminated with only PCBs (Kanechlor 400), subsequent analysis revealed that there was also extensive contamination with polychlorodibenzofurans (PCDFs) and polychlorinated quaterphenyls (PCQs). Studies in animals have clearly indicated that PCDFs are more toxic than PCBs, and a number of lines of evidence support the contention that the adverse health effects suffered by the rice oil poisoning victims were a result of PCDF rather than PCB exposure. First, the concentrations of PCBs measured in patients with symptoms of Yusho were substantially lower than those observed in a number of studies of healthy, occupationally-exposed workers (Kunita et al, 1984; Masuda et al, 1985). The occupational studies have also shown workers to have undetectable or minimally-detectable concentrations of PCDFs (Kashimoto et al, 1985), while rice oil victims in Japan and Taiwan had significant concentrations of PCDFs, including a number of extremely toxic isomers. These human observations are supported by a study in which monkeys were fed PCBs with levels of PCDF and PCQ contamination similar to that of the Yusho oils (Kunita et al, 1984). Monkeys fed this diet developed dermatologic symptoms resembling those seen in patients with Yusho. Monkeys fed PCBs without PCDF/PCQ contamination, or fed PCQs alone, did not develop dermatologic lesions. Collectively these observations have led to the conclusion that the symptoms experienced by the rice oil poisoning victims were the result of PCDF rather than PCB exposure (Kuratsune, 1980; Kashimoto et al, 1981; Drill et al, 1982; Masuda et al, 1982; Masuda and Yoshimura, 1984; Chen et al, 1984; Kunita et al, 1985; Miyata et al, 1985, and Bandiera et al, 1984).

3.2 Health Effects Information from Environmental Exposure

A number of studies have attempted to measure health effects from environmental exposure to PCBs. The most common source of environmental exposure in these studies was the consumption of Lake Michigan fish (e.g. Humphrey, 1980; Fein et al, 1984; Kreiss et al, 1981; Smith, 1983), although the results from other sources of environmental exposure have also been studied (e.g. Baker et al, 1980). The results from these studies have generally been negative. Interpretation of the meaning of the few positive observations has been hampered by either the absence of a control population matched for known risk factors for the parameter studied, a strong positive correlation between PCB body burden and the body burden of other

halogenated organics, or the inability to demonstrate increased PCB body burden in the "exposed" population compared with background body burdens in "non-exposed" controls. The absence of these apparent adverse effects in studies of workers with much higher PCB exposures (see below) casts further doubt on the significance of these findings.

3.3 Health Effects Information from Occupational Exposure

With few exceptions, individuals with the largest and longest exposure to PCBs are found in the occupational setting. For this reason, the study of occupational exposure to PCBs is probably the best source of information regarding their health effects in humans.

A population of 326 workers employed at two capacitor manufacturing plants had been the most studied, and a number of reports concerning their health have appeared in the literature. Though this population had been exposed to Aroclor 1016 during the most recent two year period prior to the initial study (and to a lesser extent to Aroclor 1221), the long-term exposure was primarily to Aroclors 1242 and 1254 (Fischbein et al, 1979). Duration of exposure to PCBs was substantial, as 40% of the workers had been employed for 20 years or longer. Air levels of PCBs varied widely throughout the plants, ranging from 0.007 mg/m^3 to 11.0 mg/m^3 (Fischbein et al, 1982). Wolff et al (1982) analyzed 290 plasma samples and 61 adipose tissue samples from these workers and found plasma concentrations of lesser chlorinated PCBs ranging from 6-2350 ppb, plasma concentrations of higher chlorinated PCBs ranging from 0-546 ppb, adipose levels of lesser chlorinated PCBs of 0.6-414 ppm, and adipose levels of higher chlorinated PCBs from 1-165 ppm. Approximately half of the population had a history of dermatologic symptoms, rash being the most common. A history of non-adolescent acne, a symptom considered characteristic of PCB exposure, was reported by 12% of the workers (Fischbein et al, 1982). Edema of the upper eyelid, eye discharge, and enlargement of the Meibomian glands, common symptoms among patients with Yusho, were each found in 7% or less of the occupationally-exposed workers (Fischbein et al, 1985). Warshaw et al (1979) studied respiratory function in 243 of the workers and found that 14% had abnormal forced vital capacity. From an extensive examination of clinical chemistry parameters in the PCB-exposed worker population, Fischbein et al (1979) concluded :

"_ there was a paucity of abnormal results in the biochemical studies. Similar findings were noted in the results of the hematologic tests _ ".

Other occupationally-exposed populations have also been studied, and have been included in a recent review by Gaffey (1983) of the human health effects of PCBs. Gaffey, in surveying the human PCB literature, classified health effect observations into five categories: dermatologic findings, liver function, fat metabolism, blood and blood pressure, and symptoms, illnesses, and other

nditions. Which respect to each of these categories, the llowing observations and conclusions were noted:

Dermatological effects: Of 11 studies of PCB-exposed workers which reported dermatologic findings, dermal symptoms were noted in all 11. Correlation of dermal symptoms with blood PCB concentrations were generally poor or non-existent. However, collectively the evidence strongly suggests that chloracne may occur when PCB blood levels exceed 150-200 µg/ml.

Liver function: Some abnormality in liver function indicated by a change in one or more relevant clinical chemistry parameters, was observed in five of seven studies of occupational exposure to PCBs. [Though not mentioned by Gaffey, it should be noted that while differences in some parameters indicative of liver function were observed to be statistically significant in some of these studies, these differences were uniformly quite small. Further, with chemical-induced hepatotoxicity one would expect to find a consistent pattern of abnormalities among overlapping indices of liver function. No such consistent pattern was observed.] In no case was the abnormality associated with any measurable adverse health effect. The remaining two studies found no evidence of liver abnormalities. A ninth study found evidence of induction of drug metabolism among PCB-exposed workers (Alvares et al, 1977).

Fat metabolism: There appears to be a correlation between serum triglycerides and PCB exposure in most studies. Results concerning cholesterol are equivocal, with one study showing an increase, one a decrease, and three no change. Conflicting results have also been observed with HDL-cholesterol. Changes in fat metabolism produced by PCBs, if they exist, do not appear to be of clinical significance.

Blood and blood pressure: None of the five studies which examined blood chemistry noted abnormalities associated with PCB exposure. One study of PCB-exposed workers measured blood pressure, but found no association with PCBs.

Symptoms, illness, and other conditions: Five studies report a variety of symptoms among PCB-exposed workers. Most of these symptoms appear to be unrelated to PCB exposure. None of the reports have found significant clinical effects to be associated with PCB exposure.

3.4 Summary of Non-cancer Human Health Effects of PCBs:

PCBs appear to be of low potency in producing adverse human health effects. Among workers with demonstrated high body burdens

of PCBs, the only consistently demonstrated clinical finding is dermatological abnormalities. Though one study has found evidence for hepatic enzyme induction in humans, there is no compelling evidence for PCB-induced liver injury despite relatively high levels of exposure. A number of subjective and objective symptoms have been reported for workers exposed to PCBs, and a variety of such symptoms would be expected in examining any population. The appearance of symptoms usually does not correlate with PCB levels or exposures, and no symptom or symptom type (other than dermal) is prominent when the studies are considered collectively. Exposure to PCBs highly contaminated with PCDFs may lead to significant symptomology, but these effects appear to be due to the more toxic PCDFs.

4.0 GENOTOXIC EFFECTS OF PCBS

4.1 Bacterial Mutagenicity Studies

Wyndham and co-workers (1976) were probably the first to test several chlorinated biphenyl mixtures in a bacterial test system. In this study the TA1538 mutant strain of *S. typhimurium* developed by Ames was used as the tester strain of bacteria, and rabbit liver microsomes were apparently added to metabolize the PCBs. Contrary to the authors' report that PCBs were weakly mutagenic under study conditions, a review of their data indicates that only 4-chlorobiphenyl and Aroclor 1221 demonstrated significant activity. Interestingly, Aroclor 1221, which has a chlorine content of 1.15 chlorine atoms per molecule and is therefore largely a monochlorobiphenyl mixture, was considerably less active than 4-chlorobiphenyl. A review of the actual results provided in the report indicate that neither 2,2',5,5'-tetrachlorobiphenyl or Aroclor 1268 are mutagenic.

Subsequent attempts to demonstrate that PCBs might cause mutations in bacterial test systems have failed. In an affadavit from Dr. Safe, senior author of the Wyndham et al (1976) article, he indicates, based on his own inability to reproduce these findings, that 4-chlorobiphenyl should not be considered mutagenic. Additional assays by other scientists also discount any suggestion that PCBs or monochlorobiphenyls are mutagenic in bacterial systems. Heddle and Bruce (1977; Bruce and Heddle, 1979) tested a number of different chemicals in the TA1535, TA1537, TA98, and TA100 strains of *S. typhimurium* , both with and without an S-9 mix to provide metabolism and possible activation of the compounds tested. Aroclor 1254 was not mutagenic in this study. Similarly McMahon et al. (1979) reported results after screening a large number of compounds in a testing scheme incorporating several auxotrophs of *S. typhimurium* and *E. coli* . The only chlorinated biphenyl tested was 4-chlorobiphenyl, but it was found to be negative, i.e. it was not mutagenic, in all 10 tester strains. Based on these studies and other reports in the literature (Schoeny et al., 1979; Rinkus and Legator, 1979; EPA, 1980;

TABLE 9

Summary of Microbial Mutagenicity Test Results

Product	Tester Strain	S. typhimurium										E. coli	
		C3076	D3052	G46	TA98	TA100	TA1000	TA1535	TA1536	TA1537	TA1538	WP2	WP2uv
Aroclor 1268		-	-	-	-	-	-	-	-	-	Neg.	-	-
Aroclor 1254		Neg.	Neg.	Neg.	Neg.	Neg.	Neg.	Neg.	-	Neg.	Neg.	Neg.	Neg.
Kanechlor 500		-	-	-	Neg.	Neg.	-	-	-	-	-	Neg.	-
2,2',5-5-tetrachloro-biphenyl		-	-	-	Neg.	Neg.	-	Neg.	-	Neg.	Neg.	-	-
[illegible]nichlor		-	-	-	Neg.	Neg.	-	Neg.	Neg.	Neg.	Neg.	Neg.	-
Aroclor 1221		-	-	-	-	-	-	-	-	-	Neg.*	-	-
4-chloro-biphenyl		-	-	-	-	-	-	-	-	-	Neg.	-	-

Source: Levinskas (1981)
* Based on re-evaluation of Wyndham et al.(1975) by Dr. Safe.

Levinskas, 1981), all of which have been summarized in Table 9, it is concluded that PCBs are not mutagenic in bacterial test systems.

4.2 Clastogenic/Chromosomal Studies

Several test systems designed to measure chromosomal damage have been used to test the potential genotoxicity of PCBs. The only human cell line tested were human lymphocytes in the study of Hoopingarner et al. (1972). The test system used phytohemagglutin stimulated cells which were treated with 100 ppm Aroclor 1254 for the first 24 hours, then again during the last eight hours of the S and G_2 stages of the cell cycle and during the three hours of mitosis. Cytological examinations of the cells during these stages of the cell cycle failed to reveal any effect of PCB treatment on chromosomal integrity.

Green et al. (1975a) tested Aroclors 1242 and 1254 for their

ability to induce chromosomal damage in the bone marrow and sperm cells of rats. Aroclor 1242 was administered either in single doses of 1250, 2500 and 5000 mg/kg, or as four daily doses of 500 mg/kg. Aroclor 1254 was tested using three different dosage regimens, i.e. five daily doses of either 75, 150, or 300 mg/kg. The two highest doses of Aroclor 1254 and all doses of Aroclor 1242 caused a loss of body weight in these animals and the multiple doses of Aroclor 1242 killed half of the test animals. Even though considerable systemic toxicity was observed at these doses, neither Aroclor was found to induce chromosomal damage in sperm cells or bone marrow cells. Dikshith et al. (1975) similarly reported that PCBs do not produce chromosomal abnormalities in the sperm cells of rats exposed to seven daily dosages of 50 mg/kg of Aroclor 1254. The results of these two studies were later reproduced and substantiated by Garthoff et al. (1977). Male rats were fed dietary levels of 5, 50 or 500 ppm of Aroclor 1254 for 5 weeks after which sperm and bone marrow cells were examined for chromosomal damage; no evidence of chromosomal damage to either cell line was obtained.

Several investigators have employed the micronucleus test as a measure of chromosomal damage. Studies using this test system have reported that PCBs do not induce micronuclei in mice injected for five consecutive days with Aroclor 1254 at doses approximating one half of the LD_{50} or at doses representing some fraction thereof (Heddle and Bruce, 1977; Bruce and Heddle, 1979; Jenssen and Ramel, 1980). The doses used in these studies also failed to induce sperm abnormalities believed to be indicative of mutations or chromosomal deletions.

Using Drosophila melanogaster and Bombyx mori as the eukaryote test species, neither of the French PCB mixtures, Clophen 30 or Clophen 50 produced chromosomal aberrations (Nilsson and Ramel, 1974).

In conjunction with a study measuring the effects of PCBs on the breeding success of Ring Doves fed a 10-ppm diet of Aroclor 1254, Peakall et al. (1972) also reported the incidence of chromosomal aberrations in eggs of the birds. The average aberration rate changed from 0.8% in controls to 1.8% in the PCB treated birds. However, the average rate of chromosomal aberrations measured in the eggs of PCB pretreated birds was only higher than the highest control value in 4/17 eggs and the authors indicated that these results were inconclusive. By comparison, Aroclor 1242 has been injected into the eggs of white Leghorn chickens until PCB concentrations reached 20 ppm (Blazak and Marcun, 1975). Even though concentrations were clearly toxic, evidence of chromosomal aberrations was not observed.

4.3 Dominant Lethal tests

Green et al. (1975b) reported that Aroclor 1242 administered as single doses of 625, 1250 and 2500 mg/kg or after five daily dosages of 125 or 250 mg/kg failed to induce dominant lethal

mutations in the rat. Aroclor 1254 administered for five days at dosages of 75, 150 or 300 mg/kg or at dietary levels of 25 or 100 ppm for 70 days was likewise without effect. Keplinger et al. (1972) and Calandra et al. (1976) tested Aroclors 1242, 1254 and 1260 in mice at dosages of 500 or 1000 mg/kg and also found them to be without dominant lethal effects.

4.4 DNA Damage Studies

On the basis of sedimentation rates, Stadnicki et al. (1979) have reported that the epoxide of tetrachlorobiphenyl caused single-stranded chromosomal breaks in the DNA of L-929 cells at concentrations ranging from 1 ug/ml to 100 ug/ml. A mixture of two hydroxylated metabolites, and to a much lesser extent tetrachlorobiphenyl, caused some damage at 20 ug/ml and what was reported as significant damage at 100 ug/ml. The significance of this single *in vitro* test is questionable, given the fact that the epoxide metabolite was the only chemical species demonstrating a strong activity in this test system. This conclusion has been reached in part because the authors, on the basis of this study, similarly concluded that the epoxide metabolite is the only chemical species of interest with regards to its potential carcinogenicity. However, all mammalian tests as well those *in vitro* tests containing some activation system were negative, indicating that either the epoxide is not genotoxic in other test systems or quantities of the epoxide sufficient to produce genotoxicity are not generated *in vivo*.

4.5 Cell Transformation Studies

Norback et al. (1981) reported that Aroclor 1254 transformed C3H10T11/2 cells to Type III foci after six weeks of continuous exposure to 10 ug/ml of Aroclor 1254, while a concentration of 1 ug/ml did not. The authors suggested that these results indicate that the effects of PCBs in cell culture may include promotion. In contrast to this study, Pienta (1980) reported Aroclor 1254 did not induce cell transformations after eight days of exposure when utilizing Syrian hamster embryo cells. The highest dose tested was 50 µg/ml, which was five times the highest dose later used by Norback et al. (1981).

4.6 Summary

The results of the preceding studies have been summarized on the next page in Table 10. Given the fact that the only test reported as positive is one of questionable significance, the number of times PCBs have been tested and found to be without significant genotoxicity lead inevitably to the conclusion that these compounds should be considered to be without evidence of genotoxic activity.

Table 10

Species	Type of Genotoxic Activity: DNA damage	Mutation	Chromosomal damage	Cell Transformation
Prokaryotes		Neg.(28/12)	Neg.(2/2)	
Mammalian cells (in vitro)	Pos.?(1/1)		Neg.(6/9)	Neg. (2/2)
Mammals (in vivo)			Neg.(3/3)	
Human cells (in vitro)			Neg.(1/1)	

The first number in the parentheses indicates the total number of times an Aroclor was tested, the second number in parentheses indicates the total number of times a specific test strain or cell line was tested. Thus, the results followed by the larger numbers in parentheses represent the results most likely to be reproduced : further testing is performed in the future.

5.0 Summary of Data Reviewed and Evaluation

5.1 Evaluation of Animal Evidence for Carcinogenicity of PCBs

The investigation of PCBs' carcinogenic potential in mice is limited to two short studies, while some eight to ten studies have been reported using various strains of rat. The PCB mixtures tested thus far are : 1) in mice - Kanechlor 300, Kanechlor 400, Kanechlor 500 and Aroclor 1254; and 2) in rats - Kanechlor 300, Kanechlor 400, Kanechlor 500, Clophen A30, Clophen A60, Aroclor 1254 and Aroclor 1260.

PCBs have been found to be tumorigenic in mice with Aroclor 1254 producing hepatomas after 11 months of exposure and Kanechlor 500 (similar in composition to Aroclor 1254) inducing hepatocellular carcinomas after 8 months of exposure. These lesions were shown to be reversible and specific for the dose (500 ppm) and chlorination of the PCB mixture.

In rats, Aroclor 1260 or its equivalent, Clophen A60, have produced hepatocellular carcinomas in three studies at doses of approximately 100 ppm, a dose which appears to represent the maximally tolerated dose for rats. A review of these three studies indicates that the tumors occur very late in the life of the animal, with a significant incidence of tumors only beginning to appear after about two years of exposure. Of interest is the fact that in all three studies the PCB treatment, while increasing the

incidence of liver cancer, did not increase the total tumor incidence. The total tumor incidence was not increased in these studies because in each case the incidence of other tumor types had been significantly decreased. This suggestion of antitumor activity of PCBs has also been demonstrated in a study examining the effect of PCB exposure on the final tumor incidence in animals following the transplantation of the Walker 256 sarcoma. The effects of chronic PCB-treatment was not life-shortening, and in fact in two of the studies the morbidity and mortality of the animals was actually decreased by PCB treatment. Furthermore, while the tumors are described as malignant, i.e. hepatocellular carcinomas, in none of the three studies did the liver tumors metastasize to other organs even though metastases would be expected if the tumors were malignant. So from these studies it is evident that PCB-treatment does not increase the total cancer risk in these animals, rather it shifts the incidence of the type of tumors observed by significantly decreasing some tumor types while enhancing the incidence of liver tumors. Lastly, PCB mixtures of lesser chlorination, i.e. Aroclor 1254 and Clophen A30 (similar in composition to Aroclor 1242, see table 1.4, page 8, of Brinkman and DeKok, 1980), have been examined in two separate studies and found not to be carcinogenic. Thus, conclusions to be drawn from the rat data, like the mouse data, are specific for the dose and degree of chlorination of the PCB mixture being tested.

Possibly because PCBs produce liver hypertrophy, they enhance the tumorigenesis of certain liver carcinogens if given after the carcinogen in question has had an opportunity to initiate tumors. However, if the PCB exposure precedes or is concomitant with exposure to liver carcinogens the tumorigenic response is typically decreased, probably as a result of an enhanced metabolic detoxification of the carcinogen.

To summarize, the qualitative human relevance of the carcinogenic activity of PCBs based on the animal data is limited. The studies providing some evidence of its carcinogenic activity are specific for the degree of chlorination of the PCB mixture, the total tumor incidence is not increased, and the tumors produced occur only very late in the life of the animal and have no adverse effect on the morbidity or mortality of the animal. There are other considerations that limit concern for the carcinogenic data in rats as well. PCBs are not mutagenic, and the mechanism of tumorigenesis for these compounds therefore would appear to involve an epigenetic mechanism. There is also substantial evidence that the doses used to induce tumors in rats are hepatotoxic, and evidence indicating the neoplasms induced by PCBs are reversible if the exposure is terminated before the animal has been exposed for a considerable portion of the animal's lifespan. All of these findings seriously undermine the human relevance of the animal carcinogenicity data. This is particularly true as the human dosages from past and present human exposures are far lower than those used in the animal studies. Given these considerations, it is concluded that the animal data provides *sufficient evidence* of *limited human relevance* that PCBs of 60% chlorine content (e.g. Aroclor 1260/Clophen A60)

are carcinogenic in animals. For PCBs having a chlorine composition of 54% (e.g. Aroclor 1254/Kanechlor 500) there is only *inadequate evidence* of carcinogenicity in animals because the larger and longer study in rats was negative, a finding suggesting that the reversible effects reported in mice may have resulted from a promotion of the substantial background incidence of liver tumors occurring in this species. For the remaining commercial PCB mixtures (i.e. Aroclor 1248, Aroclor 1242, Kanechlor 400, Kanechlor 300, and Clophen A30) there is either *no evidence* or *insufficient evidence* for the carcinogenicity of these mixtures in animals.

5.2 Evaluation of Epidemiological Evidence for Carcinogenicity of PCBs

Evidence for human carcinogenicity would be supplied if the two major epidemiological studies of PCB-exposed workers demonstrated a consistent increase in cancer mortality. This increase should show a positive correlation with exposure and evidence of latency. The evidence for carcinogenicity would be strengthened if the incidence of specific types or sites of neoplasms were consistently elevated. In reviewing the data from these two cohorts, none of these conditions are met.

While Bertazzi et al. (1986), in their study of Italian PCB-exposed workers, found a statistically significant elevation in the rate of cancer mortality among both male and female workers, the much larger study of Brown found no increase in cancer mortality. In the Brown study, there were higher-than-expected incidences of rectal and liver cancer. Evidence for an association between PCB exposure and these malignancy types cannot be considered strong, however, in that: 1) no cases of rectal cancer were observed after the initial report, suggesting that this increased rate was anomalous, 2) the number of cases of liver cancer observed in this study is not appreciably greater than expected when examined without total number of liver neoplasms used by Brown which includes those liver cancers that have metastasized from other organs, 3) in comparison, the study by Bertazzi et al. found only one case of liver cancer and no cases of rectal cancer in their cohort, and 4) the absence of a clear association with latency or relationship with duration of exposure.

The primary sites of neoplasms contributing to the higher-than-expected cancer mortality rates in the Bertazzi studies were located in the digestive system and the hematopoietic and lymphatic systems. These were not increased in the larger cohort reported by Brown. Further, in the Bertazzi cohort, there was no evidence of latency or relationship between cancer mortality and exposure to PCBs. It should also be noted that in the larger of the two subdivisions of their cohort, the female workers, differences in incidences of causes of death presumably unrelated to PCB exposure (viz., increases in accidental death and decreases in deaths from cardiovascular disease) were of similar magnitude as increases in death from malignant tumors. This suggests, at least

r this group, that other confounding variables may exist.

In summary, epidemiological evidence for human carcinogenicity of PCBs is at present weak and mostly negative. As such, until larger epidemiological studies can be completed, the data must be considered *inadequate to characterize PCBs as human carcinogens*.

5.3 Evaluation

Aroclor 1260/Clophen A60 :

There is *sufficient evidence* of *limited human relevance* for the carcinogenicity of Aroclor 1260 in animals.

The human evidence for carcinogenicity of this compound is *inadequate* .

Aroclor 1254/Kanechlor 500 :

There is *inadequate evidence* for the carcinogenicity of Aroclor 1254 in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

Aroclor 1248/Kanechlor 400 :

There is *no evidence/insufficient evidence* for the carcinogenicity of Aroclor 1248 in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

Clophen A30/Kanechlor 300/Aroclor 1242 :

There is either *no evidence* or *insufficient evidence* for the carcinogenicity of these mixtures in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

REFERENCES

Alvares, A.P. et al. 1977. Alterations in drug metabolism in workers exposed to polychlorinated biphenyls. Clin. Pharmacol. Ther. 22:140.

Anderson, L.M. et al. 1983. Effects of polychlorinated biphenyls on lung and liver tumors initiated in suckling mice by N-nirtosodimethylamine. J. Natl. Cancer Inst. 71:157.

Arai, M. et al. 1983. Comparative enhancing effects of polychlorinated biphenyls and phenobarbiyal on dimethylnitrosamine-induced hepatic and renal tumorigenesis in rats. In: Developments in the Science and Practice of Toxicology, A.W. Hayes, R.C. Schnell and T.S Miya (eds.), pp 359-362, Elsevier Science Publishers.

Bahn, A.K. et al. 1976. Melanoma after exposure to PCBs. New Engl. J. Med. 295:450.

Baker, E.L. et al. 1980. Metabolic consequences of exposure to polychlorinated biphenyls (PCB) in sewage sludge. Am. J. Epidemiology 112:553.

Bandiera, S. et al. 1984. Comparative toxicities of the polychlorinated dibenzofuran (PCDF) and biphenyl (PCB) mixture which persists in Yusho victims. Chemosphere 13(4):507.

Berry, D.L. et al. 1978. Lack of tumor-promoting ability of certain environmental chemicals in a two-stage mouse skin tumorigenesis assay. Res. Commun. Chem. Pathol. Pharmacol. 20:101.

Bertazzi, P.A. et al. In press. Cancer mortality of electrical workers exposed to PCB's. Am. J. Ind. Med.

Bertazzi, P.A. et al. 1981. Mortality study of male and female workers exposed to PCBs, presented at the International Symposium on Prevention of Occupational Cancer Helsinki, Finland.

Brinkman, U.A. Th. and A. DeKok. 1980. Production, properties and usage. In: Halogenated Biphenyls, Terphenyls, Naphthalenes, Dibenxodioxins and Related Products (ed. R.D. Kimbrough) Elsevier, New York, NY, p. 1-40.

Brown, D.P. 1986. Mortality of workers exposed to polychlorinated biphenyls - An Update. (under revision)

Brown, D.P. and M. Jones. 1981. Mortality and industrial hygiene study of workers exposed to polychlorinated biphenyls. Arch. Environ. Health. 36:120.

Calandra, J.C. 1976. Summary of toxicological studies on commercial PCBs. In: Proceedings of the National Conference on

Polychlorinated Biphenyls. USEPA report 560/6-75-004.

Chen, P.H.S. et al. 1984. Polychlorinated biphenyls, dibenzofurans, and quaterphenyls in the toxic rice-bran oil and PCBs in the blood of patients with PCB poisoning in Taiwan. Am. J. Ind. Med. 5:133.

Deml, E. et al. 1983. Benzo(a)pyrene initiates enzyme-altered islands in the liver of adult rats following single pretreatment and promotion with polychlorinated biphenyls. Cancer Lett. 19:301.

Deml, E. and D. Oesterle. 1987. Dose-reponse of promotion by polychlorinated biphenyls and chloroform in rat liver foci bioassay. Arch. Toxicol. 60:209.

DiGiovanni, J. et al. 1977. Tumor-initiating ability of 2,3,7,8-tetrachlorodibenzo-p-dioxin and Aroclor 1254 in the two stage system of mouse skin carcinogenesis. Bull. Environ. Contam. Toxicol. 18:552.

Drill, V.A. et al. 1982. Comments and Studies on the use of Polychlorinated Biphenyls in Response to an order of the United States Court of Appeals for the District of Columbia. Drill, Freiss, Hays, Loomis and Shaffer Inc., Consultants in Toxicology, Arlington, VA.

EPA (Environmental Protection Agency). 1980. Ambient Water Quality Criteria for Polychlorinated Biphenyls. EPA 440/5-80-068, PB81-117798.

Fein, G.G. et al. 1984. Prenatal exposure to polychlorinated biphenyls: Effects on birth size and gestational age. J. Ped. 105(2):315.

Fischbein, A. et al. 1979. Clinical findings among PCB-exposed capacitor workers. N.Y. Acad. Sci. 320:703.

Fischbein, A. et al. 1982. Dermatological findings in capacitor manufacturing workers exposed to dielectric fluids containing polychlorinated biphenyls (PCBs). Arch. Environ. Health 37:69.

Fischbein, A. et al. 1985. Oculodermatological findings in workers with occupational exposure to polychlorinated biphenyls (PCBs). Brit. J. Indust. Med. 42:426.

Gaffey, W.R. 1983. The epidemiology of PCBs. In: PCBS: Human and Environmental Hazards. F.M. D'Itri and M.A. Kamrin [Eds.], Butterworth Publ., Woburn, MA.

Gans, J.H. and S.J. Pintauro. 1986. Liver scarring induced by polychlorinated biphenyl administration to mice previously treated with diethylnitrosamine. Proc.Soc. Exp. Biol. Med. 183:207.

Greim, H. et al. 1985. Studies on the evaluation of tumor

promoting agents in human hepatocarcinogenesis., In: Hepatology : A Festschrift for Hans Popper, H. Brunner and H. Thaler (eds), Raven Press, New York.

Hayes, M.A. et al. 1985. Influence of cell proliferation on initiating activity of pure polychlorinated biphenyls and complex mixtures in resistant hepatocyte *in vivo* assays for carcinogenicity. J. Natl. Cancer Instit. 74:1037.

Hendricks, J.D. et al. 1980. Null effect of dietary Aroclor 1254 on hepatocellular carcinoma incidence in Rainbow trout (Salmo gairdneri) exposed to aflatoxin B_1 as embryos. J. Environ. Pathol. Toxicol. 4:9.

Hendricks, J.D. et al. 1977. Inhibitory effect of a polychlorinated biphenyl (Aroclor 1254) on aflatoxin B1 carcinogenesis in rainbow trout. J. Natl. Cancer Instit. 59:1545.

Higuchi, K. 1976. PCB Poisoning and Pollution, Academic Press, New York, N.Y.

Humphrey, H.E.B. 1980. Evaluation of humans exposed to halogenated biphenyls. Am. Chem. Soc. Div. Environ. Chem. Preprints 20(2):272.

Ito, N et al. 1978. Enhancing effect of chemicals on production of hyperplastic liver nodules induced by N-2-fluorenylacetamide in hepatectomized rats. Cancer Res. 41:3071.

Ito, N. et al. 1974. Histopathological studies on liver tumorigenesis in rats treated with polychlorinated biphenyls. Gann 65:545.

Ito, N. et al. 1973a. Histopathologic studies on liver tumorigenesis induced in mice by technical polychlorinated biphenyls and its promoting effects on liver tumors induced by benzene hexachloride. J. Natl. Cancer. Inst. 51:1637.

Ito, N. et al. 1973b. Interactions of liver tumorigenesis in mice treated with technical polychlorinated biphenyls (PCBs) and benzene hexachloride. In: New Methods in Environmental Chemistry and Toxicology, p. 141.

Kashimoto, T. et al. 1981. Role of polychlorinated dibenzofuran in Yusho (PCB poisoning). Arch. Environ. Health. 36:321.

Kashimoto, T. et al. 1985. PCBs, PCQs and PCDFs in blood of Yusho and Yu-Cheng patients. Environ. Health Persp. 59:73.

Kerklivet, N.I. and D.J. Kimeldorf. 1977a. Antitumor activity of a polychlorinated biphenyl mixture, Aroclor 1254, in rats innoculated with Walker 256 carcinosarcoma cells. J. Natl. Cancer Instit. 59:951.

Kerklivet, N.I. and D.J. Kimeldorf. 1977b. Inhibition of tumor

rowth in rats by feeding a polychlorinated biphenyl, Aroclor 1254. Bull. Environ. Contam. Toxicol. 18:243.

Kimbrough, R.D. and R.E. Linder, 1974. Induction of adenofibrosis and hepatomas of the liver in Balb/cJ mice by polychlorinated biphenyls (Aroclor 1254). J. Natl. Cancer Institt. 53:547.

Kimbrough, R.D. et al. 1975. Induction of liver tumors in Sherman strain rats by polychlorinated biphenyl Aroclor 1260. J. Natl. Cancer Inst. 55:1453.

Kimura, N.T. et al. 1976. Polychlorinated biphenyls as a promoter in experimental hepatocarcinogenesis. Z. Krebsforsch. Klin. Onkol. 87:257.

Kimura, N.T. and T. Baba, 1973. Neoplastic changes in the rat liver induced by polychlorinated biphenyl. Gann 64:105.

Kreiss, K. et al. 1981. Association of blood pressure and polychlorinated biphenyls. J. Am. Med. Assoc. 245:2505.

Kunita, N. et al. 1984. Causal Agents of Yusho. Am. J. Ind. Med. 5:45.

Kunita, N. et al. 1985. Biological effect of PCBs, PCQs and PCDFs present in the oil causing Yusho and Yu-Cheng. Environ. Health Persp. 59:79.

Kuratsune, M. 1980. Yusho. In: Halogenated Biphenyls, Triphenyls, Napthalenes, Dibenzodioxins and Related Products, R. Kimbrough (ed.), Elsevier/North Holland, New York, NY, p. 287.

Levinskas, G. 1981. "A review and evaluation of carcinogenicity studies in mice and rats and mutagenicity studies with polychlorinated biphenyls." Monsanto publications.

Loury, D.J. and J.L. Byard. 1983. Aroclor 1254 pretreatment enhances the DNA repair response to amino acid pyrolysate mutagens in primary cultures of rat hepatocytes. Cancer Lett. 20:283.

Lu, Y.C. and P.N. Wong. 1984. Dermatological, medical, and laboratory findings of patients in Taiwan and their treatments. Am. J. Ind. Med. 5:81.

Makiura, S. et al. 1974. Inhibitory effect polychlorinated biphenyls on liver tumorigenesis in rats treated with 3'-methyl-4-dimethylaminoazobenezene, N-2-fluorenylacetamide and diethylnitrosamine. J. Natl. Cancer Instit. 53:1253.

Masuda, Y. et al. 1982. Comparison of causal agents in Taiwan and Fukuoda PCB poisonings. Chemosphere 11:199.

Masuda, Y. and H. Yoshimura. 1984. Polychlorinated biphenyls and dibenzofurans in patients with Yusho and their toxicological

significance. Am. J. Ind. Med. 5:31.

Masuda, Y. et al. 1985. PCB and PCDF congeners in the blood and tissues of Yusho and Yu-Cheng patients. Environ. Health Persp. 59:53.

Miyata, H. et al. 1985. PCBs, PCQs and PCDFs in tissues of Yusho and Yu-Cheng patients. Environ. Health Perspect. 59:67.

Morgan, R.W. et al. 1981. Aroclor 1254-induced intestinal metaplasia and adenocarcinoma in the glandular stomach of F344 rats. Cancer Res. 41:5052.

Nagasaki, H. et al. 1972. Hepatocarcinogenicity of polychlorinated biphenyls in mice. Gann 63:805.

NCI (National Cancer Institute). 1978. Bioassay of Aroclor 1254 for Possible Carcinogenicity. DHEW publication No. (NIH) 78-838.

Nishizumi, M. 1980. Reduction of diethylnitrosamine-induced hepatoma in rats exposed to polychlorinated biphenyls through their dams. Gann 71:910.

Nishizumi, M. 1976. Enhancement of diethylnitrosamine hepatocarcinogenesis in rats by exposure to polychlorinated biphenyls or phenobarbital. Cancer Lett. 2:11.

Norback, D.H. and R.H. Weltman. 1985. Polychlorinated biphenyl induction of hepatocellular carcinoma in the Sprague-Dawley rat. Environ. Health Persp. 60:97.

Okumura, M. 1984. Past and current medical states of Yusho patients. In: Progress in Clinical and Biological Research, Vol.. 137, Alan R. Liss, Inc., New York.

Osterle, D. and E. Deml. 1984. Dose-dependent promoting effect of polychlorinated biphenyls on enzyme-altered islands in liver of adult and weanling rats. Carcinogenesis 5:351.

Pereira, M.A. et al. 1982. Promotion by polychlorinated biphenyls of enzyme-altered foci in rat liver. Cancer Lett. 15:185.

Preston, B.D. et al. 1981. Promoting effects of polychlorinated biphenyls (Aroclor 1254) and dibenzofuran-free Aroclor 1254 on diethylnitrosoamine-induced tumorigenesis in the rat. J. Natl. Cancer Instit. 66:509.

Schaeffer, E. et al. 1984. Pathology of chronic polychlorinated biphenyl (PCB) feeding in rats. Toxicol. Appl. Pharmacol. 75:278.

Stott, W.T. and R.O. Sinnuber. 1978. Trout hepatic enzyme activation of aflatoxin B1 in a mutagen assay system and the inhibitory effect of PCBs. Bull. Environ. Contam. Toxicol. 19:35.

Shelton, D.W. et al. 1984a. Effect of dose on the inhibition of carcinogenesis/mutagenesis by Aroclor 1254 in Rainbow trout fed aflatoxin B1. J. Toxicol. Environ. Health 13:649.

Shelton, D.W. et al. 1984b. The hepatocarcinogenicity of diethylnitrosamine to Rainbow trout and its enhancement by Atoclors 1242 and 1254. Toxicol. Lett. 22:27.

Ward, J.M. 1985. Proliferative lesions of the glandular stomach and liver in F344 rats fed diets containing Aroclor 1254. Environ. Health Persp. 60:89.

Warshaw, R. et al. 1979. Decrease in vital capacity in PCB-exposed workers in a capacitor manufacturing facility. N.Y. Acad. Sci. 320:277.

Wolff, M.S. 1982. Body burden of polychlorinated biphenyls among persons employed in capacitor manufacturing. Int. Arch. Occup. Environ. Health. 49:199.

Young, S.S. 1985. Letter to the Editor. Toxicol. Appl. Pharmcol. 78:321.

LIST OF TABLES

EXPOSURE ASSESSMENT FOR SMALL BURROWING ANIMALS

Daily intake of PCBs (mg/kg/day) (I)

PCBs are taken in through the skin, the lungs, and the mouth.

$I = I_D + I_I + I_O$

I_D = Dermal intake
I_I = Inhalation intake
I_O = Oral intake

Dermal uptake is a function of the efficiency of absorption through the skin, the surface area exposed, and the PCBs in the soil covering the skin.

$I_D = E_D S L C_S$

$= 0.05/day \times \frac{36cm^2}{0.03kg} \times 10^{-6}kg/cm^2 \times C_S$

$= 0.00006/day\ C_S$

$= 0.00006/day\ C_S$

E_D = Efficiency of absorption of PCBs in soil through the skin
= 0.05/day (Hwang et al 1986 p.12-16)

S = Surface area exposed on a 30g vole

= 36 cm^2/30g/day. Includes ear pinnae, legs, underbelly, face and tail

L = Soil layer, assumed = 1.0 mg/cm^2
= $10^{-6}kg/cm^2$

C_S = Concentration in the soil

NOTE: There are no published reports of dermal uptake of PCBs in soil for voles. The rate chosen was one assumed for PCBs by Hwang (1986). Poiger and Schlatter (1979) report an uptake rate of 2.2%/day for TCDD in rats. The skin of rats is thicker than the skin of most voles, so absorption would be expected to be somewhat higher in voles.

The soil layer on the skin of the vole was estimated using the assumption of two times the soil covering found by Lepow et al (1975) on urban children (2 x 0.5116 mg/cm^2 = 1.023). This may be an underestimate for ear pinnae and other areas that are relatively naked and well vascularized but an overestimate in furry areas.

Inhalation intake

The concentration of PCBs in the vapor of the burrow was calculated using the method of MacKay (1985 and pers. comm.) assuming equal fugacities in the soil and air.

Concentration of PCBs in the air of a burrow

$K_{AW} = H/RT$

$K_{AW} = \frac{H\ (Pa\ m^3/mol)}{8.314\ Pa\ m^3/mol\ K^\circ \times T}$

$= \frac{202.6\ Pa\ m^3/mol}{(8.314\ Pa\ m^3/mol\ K^\circ \times 283.2\ K^\circ}$

Source: Mackay 1985

K_{AW} = Air water partition coefficient

T = Temperature assumed 10°C
= 283.2K°

R = 8.314 Pa m^3/mol K

H = Henry's law constant

= 2.0 x 10^{-3} atm m^3/mol 25°C (Aroclor 1254)

= 2.0 x 10^{-3} atm m^3/mol x $\frac{101325\ Pa}{atm}$

= 202.6 Pa m^3/mol

$K_{AW} = 0.0860$

$K_D = K_{OC}OC$

K_D = Soil water partition coefficient (L/kg)

OC = Fraction organic carbon (assumed = 0.02)

= 45392 x 0.02
= 907.84

K_{OC} = Soil organic carbon partitioning coefficient (following Kenaga and Goring 1980) (log K_{OC} = 0.544 log K_{OW} + 1.377)

p = Soil bulk density (assumed to be = 1.04 g/cm^3, the average in Brady (1974) (p. 53) for uncropped soil)

$K_{AS} = K_{AW}/pK_D$

K_{AS} = Air soil partitioning coefficient

= 0.0860/(907.84 x .96 L/kg)

= 9.868 x 10^{-5}kg/L

Bush et al (1986) measured the concentration in air over contaminated soil. When these data are converted to an estimate of the K_{AS}, they yield a value of 1.186 x 10^{-6}kg/L. As expected, this is below the theoretical constant at equilibrium. Even on the calm day that the measurement was taken, one can expect that equilibrium will not be attained because of some diffusion and convection away from the site.

As a worst case, consider small mammals like the pocket gopher, or mole, which spend virtually all of their time in burrows, with the burrow sealed off (Chase et al 1983); (Yates and Pedersen 1983).

Inhalation is assumed to have two phases, an active phase and a resting phase. The animal is assumed to have the same activity patterns as a pocket gopher (Chapman and Feldhamer, 1982 p. 248) and a respiration rate ratio (active: resting) equivalent to that given by Vleck (1979) for pocket gophers. It is assumed that vapor and dust are inhaled during the active phase, but only vapor is inhaled during the resting phase.

$I_I = C_S (K_{AS} (V_A + V_R) + V_A D)/W$

$= C_S (K_{AS} (RT_A M + RT_R) + RT_A MD)/W$

$= C_S (K_{AS} R (T_A M + T_R) + RTAMD)/W$

$= C_S$ (9.868 x 10^{-5}kg/L x 1.5 L/hr (9 hr/day x 4.1 + 15 hr/day) + (1.5 L/hr x 9 hr/day x 4.1 x 10^{-10}kg/L)/0.03kg

$= C_S$ x 0.256/day

V_A = Volume inhaled during active part of day = $RT_A M$ (L/day)

V_R = Volume inhaled while resting = RT_R (L/day)

R = Resting respiration rate = 1.5 L/hr

T_A = Time spent active per day = 9 hr/day

M = Active/resting rate = 4.1

T_R = Time spent resting/day = 15 hr/day

D = Dust level (assumed = 10 mg/m^3 USEPA) = 10^{-10}kg/L

W = weight = 30g = 0.03kg

Oral intake

Oral intake is a function of the ingestion rate of food, soil, and water, the concentration of PCBs in plants as a function of the concentration in soil, and the efficiency of dietary absorption.

$I_O = C_S(I_W + I_F + I_S)$

$I_W = R_W C_S/K_D$

$= 0.01\ C_S/907.84$

$= 1.1 \times 10^{-5} C_S$

I_W = Ingestion rate of PCBs in water

I_F = Ingestion rate of PCBs in food

I_S = Ingestion rate of PCBs in soil assumed to be 10% of diet

R_W = Rate of ingestion of water = 10%/day

R_F = Rate of ingestion of food = 0.2/day

$I_F = R_F BCF_P C_S$ — BCF_P = Bioconcentration factor for plants

$= 0.2/day \times 0.10\ C_S$ — = 10% (Bush et al 1986)

$= 0.02/day\ C_S$ — K_D = Soil water partitioning coefficient $C_W = C_S/K_D$

$I_S = R_S\ C_S$ — R_S = Rate of ingestion of soil (assuming 5% of diet is food)

$= .01\ C_S$ — $= .05 \times R_F = .01$

$I_O = C_S\ (1.1 \times 10^{-5}/day + 0.02/day + 0.01/day)$

$= 0.030/day\ C_S$

$I = I_D + I_I + I_O$

$= C_S\ (0.00006/day + 0.256/day + 0.030/day) \times C_S$

$= 0.286/day \times C_S$

A diet of 1 ppm PCBs has been found to increase the liver weights in F_1 male weanling rats (Linder et al 1974) and decrease the circulating levels of adrenal cortex hormone B (Byrne et al 1988). If a small mammal consumes 20 percent of its body weight per day, 1 ppm in the diet is equivalent to 0.2 mg/kg/day. At this dietary level, the soil concentration would be 0.7 mg/kg.

if 1 ppm => 1 mg/kg in food $\times \frac{0.2\ kg\ food}{kg\ body\ wt/day}$ = 0.2 mg/kg/day

if 10 ppm => 1 mg/kg $\times \frac{0.2\ kg\ food}{kg\ body\ wt/day}$ = 2 mg/kg/day

$0.2\ mg/kg/day = (0.286/day)\ C_S$

$C_S = \frac{0.2\ mg/kg/day}{0.286/day} = C_S$

$= 0.699\ mg/kg$

A diet of 10 to 20 ppm decreases the weight of reproductive organs, growth rate and reproductive success of second generation white-footed mice (Linzey 1988, Linder 1974). At this dietary level, the soil concentration would be 7 mg/kg to 14 mg/kg.

$$C_s = \frac{2 \text{ mg/kg/day}}{0.286/\text{day}}$$

$$= 6.99 \text{ mg/kg}$$

References

Brady, N.C. 1974. The Nature and Properties of Soils p.53. 8th edition MacMillan Publishing Co. New York 639 pp.

Byrne, J.J., J.P. Carbane, and M.G. Pepe. 1988. Suppression of serum adrenal cortex hormones by chronic low-dose polychlorobiphenyl or polybromobiphenyl treatments. Arch. Environ. Contam. Toxicol. 17:47-53.

Bush, B., L.A. Shane, L.R. Wilson, E.L. Barnard, and D. Barnes. 1986. Uptake of polychlorobiphenyl congeners by purple loosestrife (Lythrium salicaria) on the banks of the Hudson River. Arch. Environ. Contam. Toxicol. 15:285-290.

Chase, J.D., W.E. Howard, and J.T. Roseberry. 1983. Pocket gophers pp. 239-255 in Wild Mammals of North America. J.A. Chapman and G.A. Feldhamer, editors. Johns Hopkins University Press. Baltimore and London. 1137 pp.

Hwang, S.T., J.W. Falco, and C.H. Nauman. 1986. Development of advisory levels for polychlorinated biphenyls (PCBs) clean-up.

Kenaga, E.E., and C.A.I. Goring. 1980. Relationship between water solubility, soil sorption, octanol-water partitioning coefficient, and concentration of chemicals in biota. Aquatic Toxicology ASTM STP 707. J.G. Eaton, P.R. Parrish, and A.C. Hendricks, Eds. American Society for Testing and Materials. p.78-115.

Mackay, D. 1985. Air/water exchange coefficients. Chap. 5 in Environmental exposure from chemicals. Vol. 1. W.B. Neely and G.E. Blau (eds.) CRC Press. Boca Raton, Fla. 245 pp.

Lepow, M.L., L. Bruckman, R.A. Rubino, S. Mankowitz, M. Gillette, and J. Kapish. 1975. Investigations into the sources of lead in the environment of urban children. Environ. Res. 10:415-416.

Linder, R.E., T.B. Gaines, and R.D. Kimbrough. 1974. The effect of polychlorinated biphenyls on rat production. Fd. Cosmet. Toxicol. 12:63-77.

Linzey, A.V. 1988. Effects of chronic polychlorinated biphenyls exposure on growth and reproduction of second generation white-footed mice (Peromyscus leucopus). Arch. Environ. Contam. Toxicol. 17:39-45.

Poiger, H. and Ch. Schlatter. 1979. Influence of solvents and absorbents on dermal and intestinal absorption of TCDD. Food Cosmet. Toxicol. 18:477-481.

Vleck, D. 1979. The energy cost of burrowing by the pocket gopher Thomomys bottae. J. Physiol. Zool. 52:122-136.

Yates, T.L. and R.J. Pedersen. 1983. Moles. pp 37-51 in Wild Mammals of North America. J.A. Chapman and G.A. Feldhamer, editors. Johns Hopkins University Press. Baltimore and London. 1147 pp.

References

Brady, N.C. 1974. The Nature and Properties of Soils p.53. 8th edition MacMillan Publishing Co. New York 639 pp.

Byrne, J.J., J.P. Carbane, and M.G. Pepe. 1988. Suppression of serum adrenal cortex hormones by chronic low-dose polychlorobiphenyl or polybromobiphenyl treatments. Arch. Environ. Contam. Toxicol. 17:47-53.

Bush, B., L.A. Shane, L.R. Wilson, E.L. Barnard, and D. Barnes. 1986. Uptake of polychlorobiphenyl congeners by purple loosestrife (Lythrium salicaria) on the banks of the Hudson River. Arch. Environ. Contam. Toxicol. 15:285-290.

Chase, J.D., W.E. Howard, and J.T. Roseberry. 1983. Pocket gophers pp. 239-255 in Wild Mammals of North America. J.A. Chapman and G.A. Feldhamer, editors. Johns Hopkins University Press. Baltimore and London. 1137 pp.

Hwang, S.T., J.W. Falco, and C.H. Nauman. 1986. Development of advisory levels for polychlorinated biphenyls (PCBs) clean-up.

Kenaga, E.E., and C.A.I. Goring. 1980. Relationship between water solubility, soil sorption, octanol-water partitioning coefficient, and concentration of chemicals in biota. Aquatic Toxicology ASTM STP 707. J.G. Eaton, P.R. Parrish, and A.C. Hendricks, Eds. American Society for Testing and Materials. p.78-115.

Mackay, D. 1985. Air/water exchange coefficients. Chap. 5 in Environmental exposure from chemicals. Vol. 1. W.B. Neely and G.E. Blau (eds.) CRC Press. Boca Raton, Fla. 245 pp.

Lepow, M.L., L. Bruckman, R.A. Rubino, S. Mankowitz, M. Gillette, and J. Kapish. 1975. Investigations into the sources of lead in the environment of urban children. Environ. Res. 10:415-416.

Linder, R.E., T.B. Gaines, and R.D. Kimbrough. 1974. The effect of polychlorinated biphenyls on rat production. Fd. Cosmet. Toxicol. 12:63-77.

Linzey, A.V. 1988. Effects of chronic polychlorinated biphenyls exposure on growth and reproduction of second generation white-footed mice (Peromyscus leucopus). Arch. Environ. Contam. Toxicol. 17:39-45.

Poiger, H. and Ch. Schlatter. 1979. Influence of solvents and absorbents on dermal and intestinal absorption of TCDD. Food Cosmet. Toxicol. 18:477-481.

Vleck, D. 1979. The energy cost of burrowing by the pocket gopher Thomomys bottae. J. Physiol. Zool. 52:122-136.

Yates, T.L. and R.J. Pedersen. 1983. Moles. pp 37-51 in Wild Mammals of North America. J.A. Chapman and G.A. Feldhamer, editors. Johns Hopkins University Press. Baltimore and London. 1147 pp.

SECTION 19 - SITE 12, AREA 14 IMPOUNDMENT

19.1 Site Description

Site 12 is located within Area 14, which is a manufacturing area on the south side of the lake (See Figure 19-1). Munitions loading activities occurred in Area 14 in the past. Many of the buildings have been abandoned or demolished, but a few industries continue to operate at Area 14. The area in the vicinity of Site 12 is currently occupied by Diagraph-Bradley for the manufacture of printing inks, stencils, stencil boards, and marking pens.

Historic aerial photographs indicate apparent impoundment activity in Area 14 east of the presently occupied buildings in 1943. By 1951, the impoundment had been taken out of service and the area was partially vegetated. The aerial photographs show what appears to be an aboveground tank situated in the middle of the former impoundment area. The access road is still visible in the 1951 photographs. Photographs from 1960 still show the tank, with the surrounding area completely grown over with vegetation. The access road appears unused in these photographs. The next set of aerial photographs represent 1965 conditions and indicate that the tank has been removed.

The site is currently a circular dry impoundment with a diameter of approximately 100 feet. The interior of the impoundment is presently overgrown with trees with trunk diameters of 8 to 10 inches. The impoundment walls are about 6 feet high and the north wall was breached between 1960 and 1965, according to the air photos, to allow drainage to flow from the impoundment to an adjoining field. Several black oily pools are evident in and around the basin. Other bare patches of black sediment and tars are located around the basin floor.

19.2 Site Investigations

19.2.1 Phase I Site Investigations:

One composite soil sample (0-1 ft depth) and one composite sediment sample (0-1 ft depth) were collected. The sediment sample was later resampled for full priority pollutant analysis. The scheduled water analysis was canceled because the site was dry at the time of sampling.

19.2.2 Phase II Site Investigations:

No samples were collected in Phase II.

19.3 Analytical Results (See Appendix I, Page 12)

TOC levels in soils and sediments ranged from 12,039 to 16,673 mg/kg, while TKN concentrations of 369 to 2,267 mg/kg were detected. The results of the FID screening were 16,934 ug/kg, resulting in the collection of an additional sample for full priority pollutant analysis. The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present. N-nitrosodiphenylamine was detected in the sediment at 2,174 ug/kg wet weight. Small concentrations (less than 1 mg/kg) of other base/neutral extractable compounds (phenanthrene, pyrene, fluorene and anthracene) were quantified in the sediment but were present below the detection limits.

19.4 Environmental Effects

19.4.1 Qualitative Assessment

This site was chosen for investigation based on a history of munitions activity at this location. Historical aerial photography reveals

that the site was utilized as an impoundment as early as 1943.

Concentrations of TOC, TKN, and FID were detected in soils and sediments, but were consistent with those of the control sites. N-nitrosodiphenylamine was detected in the sediment sample, at a level higher than at any other site at the Refuge. N-nitrosodiphenylamine (NDA) has produced a carcinogenic response in laboratory tests with animals, thus, it will be considered as the site indicator contaminant in the quantitative assessment for humans and wildlife. As a conservative measure, the level of NDA detected in site sediment (2,174 ug/kg) will be considered as two times this level (4,348 ug/kg) due to deficiencies in the Phase I laboratory analysis.

In addition, trace base/neutral extractable compounds below 1 mg/kg detected in sediments included phenanthrene, pyrene, fluorene, and anthracene.

19.4.2 Quantitative Assessment

Humans

During the Phase I investigation, the only elevated contaminant of concern was N-nitrosodiphenylamine in the impoundment sediment, at a level of 2,174 ug/kg. Since the impoundment is currently overgrown with trees and is also surrounded by a berm, it is unlikely that a complete pathway exists for any human exposures to this contaminant source. However, in order to provide some framework for the upper bound risks that could be posed by chronic exposure of a site visitor to residues of this magnitude, the following scenario was analyzed.

It is assumed that a human visitor to the site might ingest an average of 100 mg of impoundment sediments as a result of direct contact

with the material. It is also assumed that the sediment contains 4,348 ug/kg of N-nitrosodiphenylamine, twice the level found in the single sediment sample analyzed, in order to provide a conservative upper bound on the residue level in the absence of more sample data. Exposure at this level would result in an average intake of 0.435 ug N-nitrosodiphenylamine per visit. Assuming a 70 kg adult such as a site inspector or Refuge employee might visit this site roughly three times yearly over a lifetime of 70 years, and using the EPA carcinogenicity potency factor of 0.005 $(mg/kg/day)^{-1}$ (Exhibit A), an excess lifetime risk of cancer of 2.5×10^{-10} is estimated. Given the upper bound assumptions employed in this assessment, and the consideration that "acceptable" risk levels are construed as levels below 10^{-4} (USEPA 540/1-85/060 1986), this represents a negligible level of risk.

Wildlife

The detection of N-nitrosodiphenylamine in a soil sample analyzed in the Phase I survey of this site also presents a mechanism for exposure for terrestrial wildlife via the direct contact route. The levels of exposure would be greatest amongst small mammals as a result of inadvertent ingestion and inhalation of contaminated soil residues and dust during daily burrowing, feeding and grooming. Thus, the risks of direct contact of these species to site nitrosamines residues were assessed. The risks to larger and/or less sensitive species, or to those which have less contact with soil residues would be lower. A search of on-line data bases (Pollution Abstracts, Biosis Previews, NTIS, HSDB) did not identify published studies on the effects of N-nitrosodiphenylamine on pertinent wildlife species. Therefore, tests with surrogate species (i.e. laboratory rodents) are used in the assessment below.

Using a breathing rate of 0.006 m3/hour for an active 30 g mouse (USEPA ECAO-CIN-477, 1985), and creation of a 10 mg/m3 dust containing 4,348 ug/kg N-nitrosodiphenylamine during one hour of daily burrowing, a daily chronic inhalation exposure of 0.0087 ug/kg/day is obtained. Exposures via the ingestion route due to consumption of contaminants in soil or vegetation are estimated at 0.041 mg/kg/day using an ingestion rate of 5% body weight or 1.5 g food/day (10% soil, and 90% vegetation containing 1% of the soil concentration). These exposures are calculated assuming levels in soil at twice the measured concentration of contaminant. Thus a total daily exposure rate of 0.041 mg/kg/day is estimated for small burrowing animals from inhalation and ingestion at this site. Using the EPA carcinogenicity unit risk factor of 0.005 $(mg/kg/day)^{-1}$ for N-nitrosodiphenylamine (Exhibit A), developed on the basis of rat studies, the total estimated daily chronic exposure would result in a risk level of 2.0×10^{-4}. The significance of this exposure is discussed below.

As discussed by Newell et al.(1987), concerns regarding the effects of cancer on wild populations are largely unknown, and risk levels of concern to humans are not directly transferable to wildlife. Many other factors come into play when addressing whether a wildlife population can maintain itself (i.e. survival to reproductive age, competition, weather, disease, predation, etc), and the effect of cancer, generally forming later in an exposed organism's lifetime, might thus be very small. On this basis, Newell et al. (1987) chose a risk level of 10^{-2} as a level of acceptable carcinogenic risk for wildlife, with the acknowledgement that more study is needed to justify this choice. Using this rationale, it is concluded that wildlife exposure to site residues of N-nitrosodiphenylamine are well below the level which could result in a carcinogenic response.

Additional review of the literature on the effects of N-nitrosodiphenylamine to wildlife is presented in the Toxicological Profile for N-nitrosodiphenylamine (ATSDR, 1987). Using the subchronic no observed adverse effect level of 150 mg/kg/day for oral exposure of laboratory rats to N-nitrosodiphenylamine derived in this review, and a safety factor of 10, a wildlife ADI of 15 mg/kg/day is derived. The estimated exposure of 0.047 mg/kg/day for burrowing rodents at this site is well within this acceptable intake level.

19.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a history of munitions activity on the site, aerial photography, site inspection, and sample analyses. A visual survey of the site included black oily pools in and around the basin. Black sediment and tars were also observed on the basin floor.

Chemical residue information consisted of analytical results for surface soil samples. The soil sample represented only the top one foot of soil; deeper soil borings and ground water monitoring were not conducted. Since contamination of the site occurred possibly through surface spillage of waste, soil and sediment contamination would most likely be found at the surface. Since there is no evidence to suggest that the surrounding soil or sediment has been excavated or otherwise disturbed, these samples should adequately represent the conditions of the site.

The quantitative risk assessment was performed under a worst case chronic exposure scenario of lifetime exposures and considered only the single data point available for a soil sample in which

N-nitrosodiphenylamine was detected in the Phase I survey. This result is only qualitatively reliable due to insufficient QA/QC supporting the analysis. In addition, the residue concentration detected was doubled in the risk calculation to provide a more conservative outcome. This worst case approach resulted in risk levels that provide an ample margin of safety for protection of potential human and wildlife receptors under the conditions assumed.

19.5 Preliminary Remedial Alternatives

Preliminary Phase I screening results discussed in the previous section indicated that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore this site was not included in the Phase II investigations. There will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

19.6 Conclusions and Recommendations

It can be concluded that the Area 14 impoundment site does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

SECTION 20 - SITE 13, AREA 14 CHANGE HOUSE

20.1 Site Description

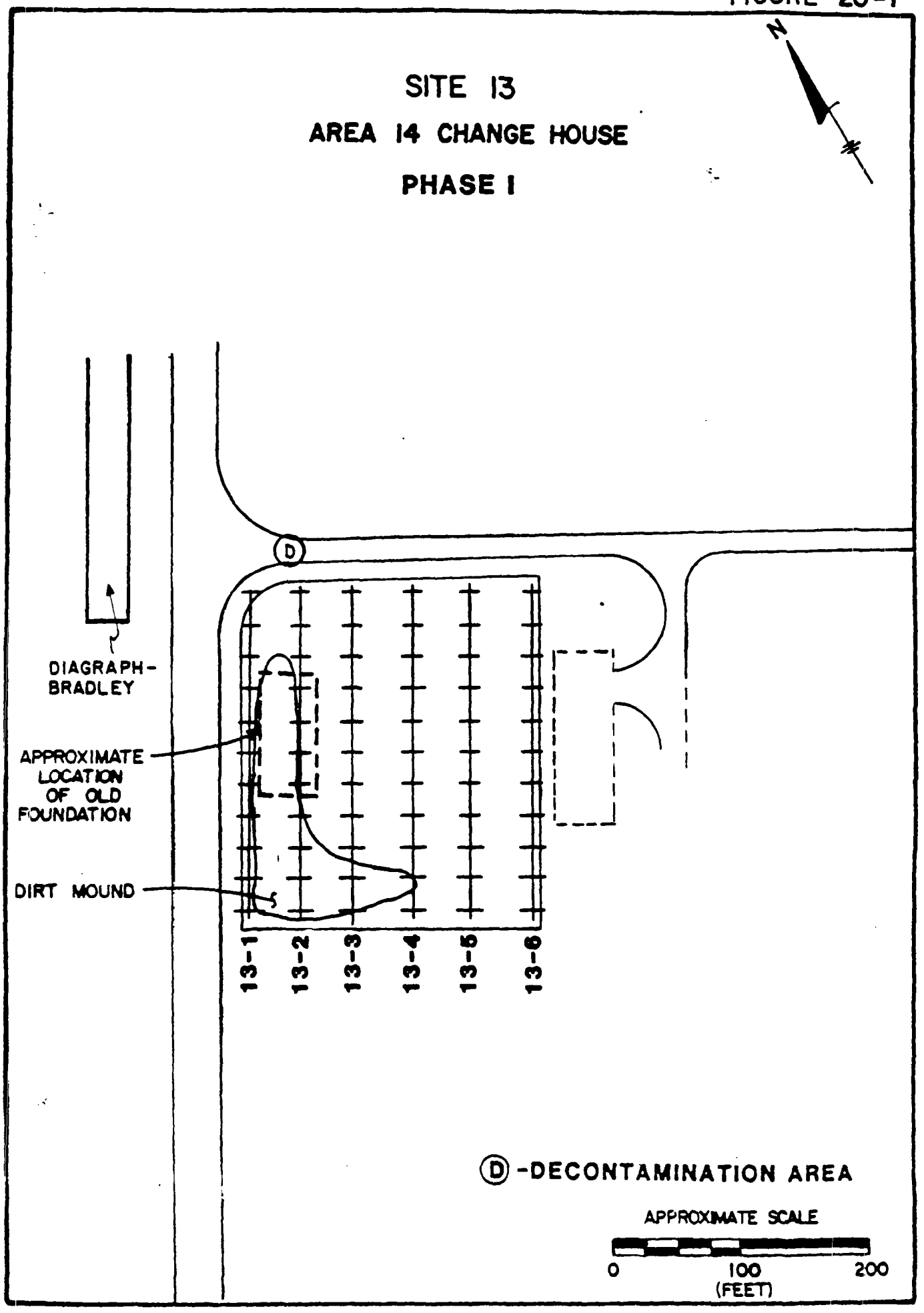
Site 13 is located southeast of the active Diagraph-Bradley buildings in Area 14 (See Figure 20-1). Further information on Area 14 can be found in Section 19.1. Site 13 consists of the site of a building which was demolished sometime between 1971 and 1980, according to aerial photographs from those years. The site is presently an open field covered with tall grass. Formerly, it was the site of a "Change House" where workers changed their clothing after working in the adjacent bomb-loading buildings. At one time a company named CTI (possibly Chemicals and Technology, Inc.) manufactured explosives and other chemicals in this building, according to the Refuge Manager. Other industries may also have occupied this building. The change building was located across from the bomb-loading building on a plot of land just southeast of the intersection of two roads on the north edge of a big dirt mound. The concrete floor of the Change House is under this mound, according to the Refuge Manager. Aerial photos show another building (no longer present) further east of the corner; field inspection revealed several half-inch reinforcing rods imbedded in concrete near the corners of the building.

20.2 Site Investigations

20.2.1 Phase I Site Investigations

A magnetometer and electromagnetic terrain conductivity survey was conducted over a 250 ft x 200 ft area, with grid spacings of 25 ft centers. The results of these surveys are shown on Figures 20-2 and 20-3. No magnetic anomalies of any significan. were noted, indicating that there is no buried metallic debris at the site.

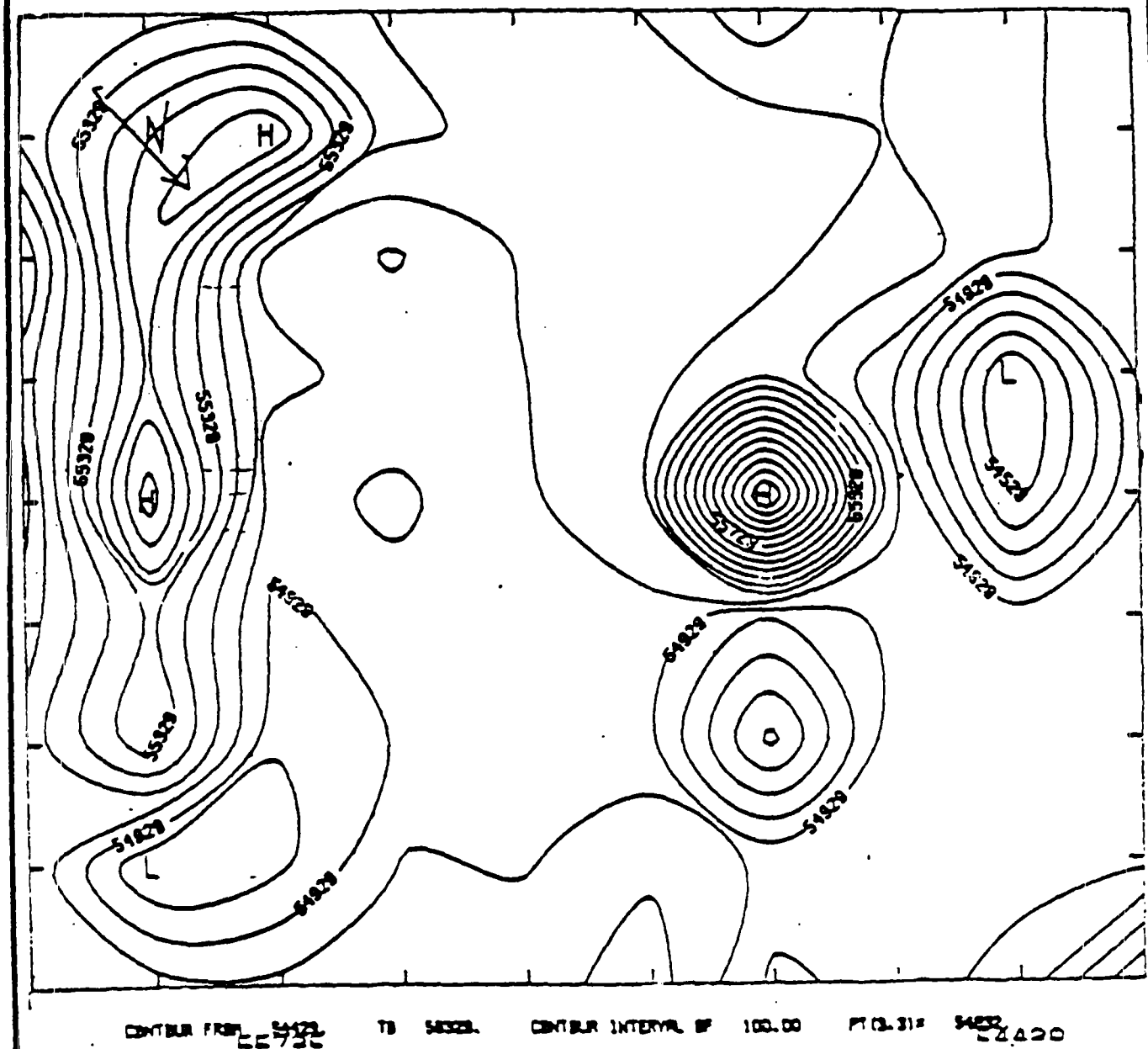
FIGURE 20-1



O'BRIEN & GERE
ENGINEERS INC

FIGURE 20-2

SITE 13
MAGNETOMETER SURVEY

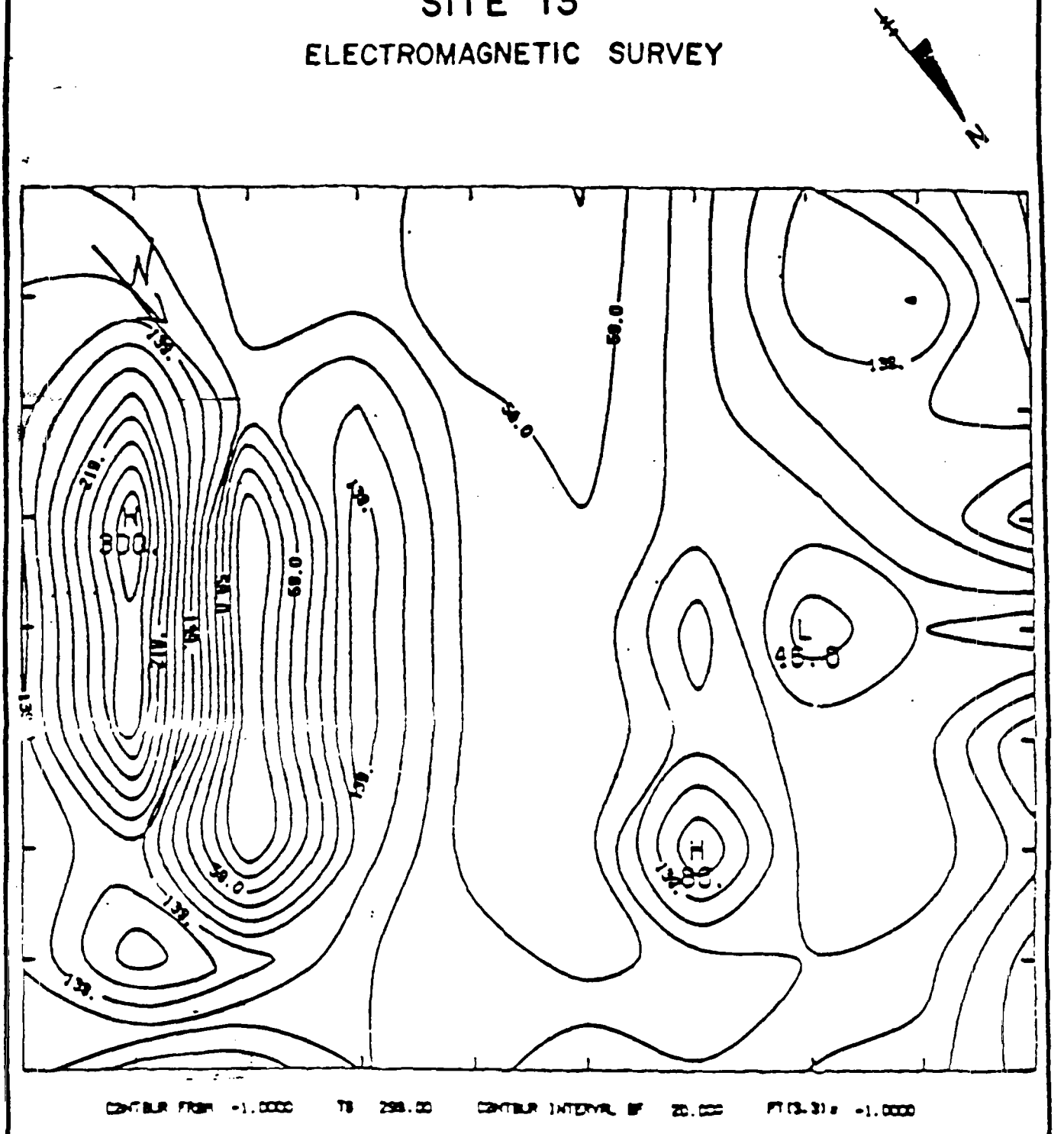


O'BRIEN & GERE ENGINEERS INC

FIGURE 20-3

SITE 13

ELECTROMAGNETIC SURVEY



Six composite soil samples (0-1 ft depth) were collected along north-south transect lines (Figure 20-1) and screened for priority pollutants, metals, cyanide, indicators and explosives.

20.2.2 Phase II Site Investigations

No additional sampling was done in Phase II.

20.3 Analytical Results (See Appendix I, page 13)

The geophysical surveys did not indicate that major buried articles are present. The soil concentrations were consistent with those detected at the control sites, although these concentrations were estimated for screening purposes only (see Exhibit B). No organic compounds were detected with the exception of delta-BHC in one soil sample (69 ug/kg), which was slightly over the detection level.

20.4 Environmental Effects

20.4.1 Qualitative Assessment

This site was chosen for investigation based on verbal accounts regarding a history of munitions activity at this location. There was, however, no history of the disposal of wastes and the historical aerial photography review as well as the geophysical surveys did not reveal the existence of a waste disposal area. The absence of a waste disposal area was further supported by the results of the Phase I sampling. On this basis, it can be concluded that there is no "source" of waste materials for on-site exposures or for migration to off-site locations.

Because there is no established waste source at this location, it is not possible to have a "complete" exposure scenario. Therefore, on the

basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

20.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

20.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was the verbal accounts of activities on the site, aerial photography, and site inspection. The verbal accounts suggested that the area could have wastes on it, although the use of the area for waste disposal is not indicated.

Chemical residue information consisted of analytical results on surface soil samples. The soils sampled represented only the top one foot of soil; deeper soil borings and ground water monitoring were not conducted. It appears that the ground was disturbed during the time that the change house was demolished. The surface soils could have been moved around or dirt hauled in to cover the building foundation. Surface sampling would be adequate for determining that there are no contaminants on the present soil surface, but it would not be adequate for identifying munitions contaminants present on the soil surface (now possibly buried) that were exposed at the time the change house was in use. Therefore no conclusions regarding subsurface conditions could be drawn exclusively on the basis of the residue information. However, the

geophysical surveys did not suggest the presence of unexplained subsurface metallic anomalies.

It can be concluded that the data generated are adequate when considered in light of the fact that there was no known history of waste disposal at this location. Residues related to the changing of uniforms soiled during munitions loading activities would have settled on the surface soils of the area. The surface soil sampling program would therefore be an adequate means of locating and identifying munitions related chemical residues.

20.5 Preliminary Remedial Alternatives

Preliminary Phase I screening results discussed in the previous section indicated that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore this site was not included in the Phase II investigations. There will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

20.6 Conclusions and Recommendations

It can be concluded that the Change House site does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

SECTION 21 - SITE 14, SOLVENT STORAGE

21.1 Site Description

Further information on Area 14 and Diagraph-Bradley is presented in Section 19.1. Linseed oil and various solvents are handled in bulk and in drums. Some of the bulk solvents noted were: T25 Xylene, T8 Diacetone Alcohol, T9 Diethylene Glycol, and T18 Methyl Cellosolve. Several compressed gas cylinders are also present. At least two drum storage areas containing 50 to 200 drums were also noted. Spill containment facilities are minimal. Site 14 is a drainage ditch adjacent to the active manufacturing operations of Diagraph-Bradley (See Figure 21-1). The ditch receives runoff from a manufacturing area where solvents are handled in bulk and in drums. The ditch runs north parallel to the road west of the buildings. Process water from the Diagraph-Bradley buildings enters this ditch from a standpipe.

21.2 Site Investigations

21.2.1 Phase I Site Investigations

Two composite water samples and two composite sediments (0-1 ft depth) were collected. One sediment composite was resampled for full priority pollutant analysis.

21.2.2 Phase II Site Investigations

One composite water sample and one composite sediment sample were collected by the drum storage area (See Figure 21-2). The Phase II samples were analyzed for purgeable and base/neutral/ acid extractable organics.

FIGURE 21-1

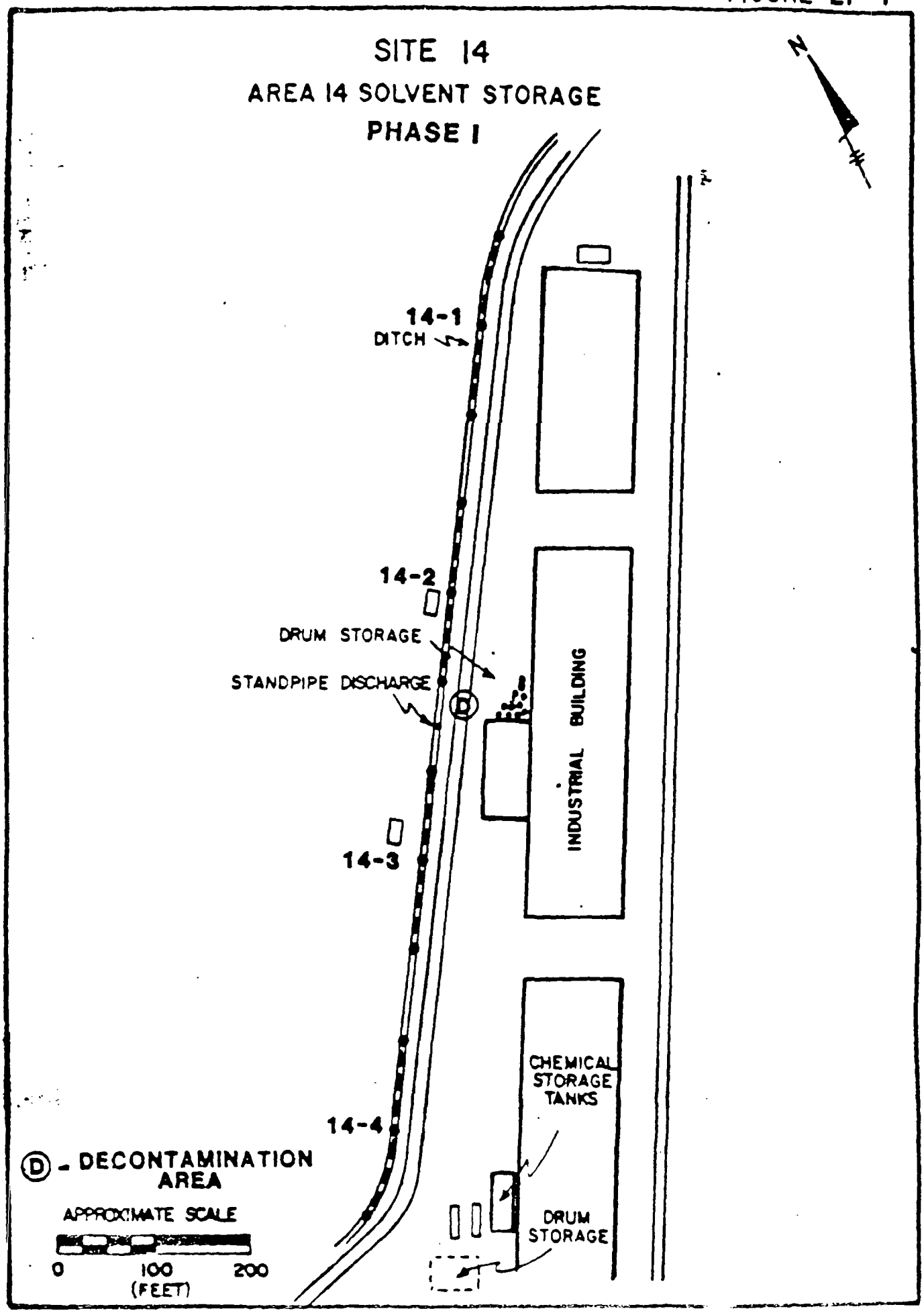
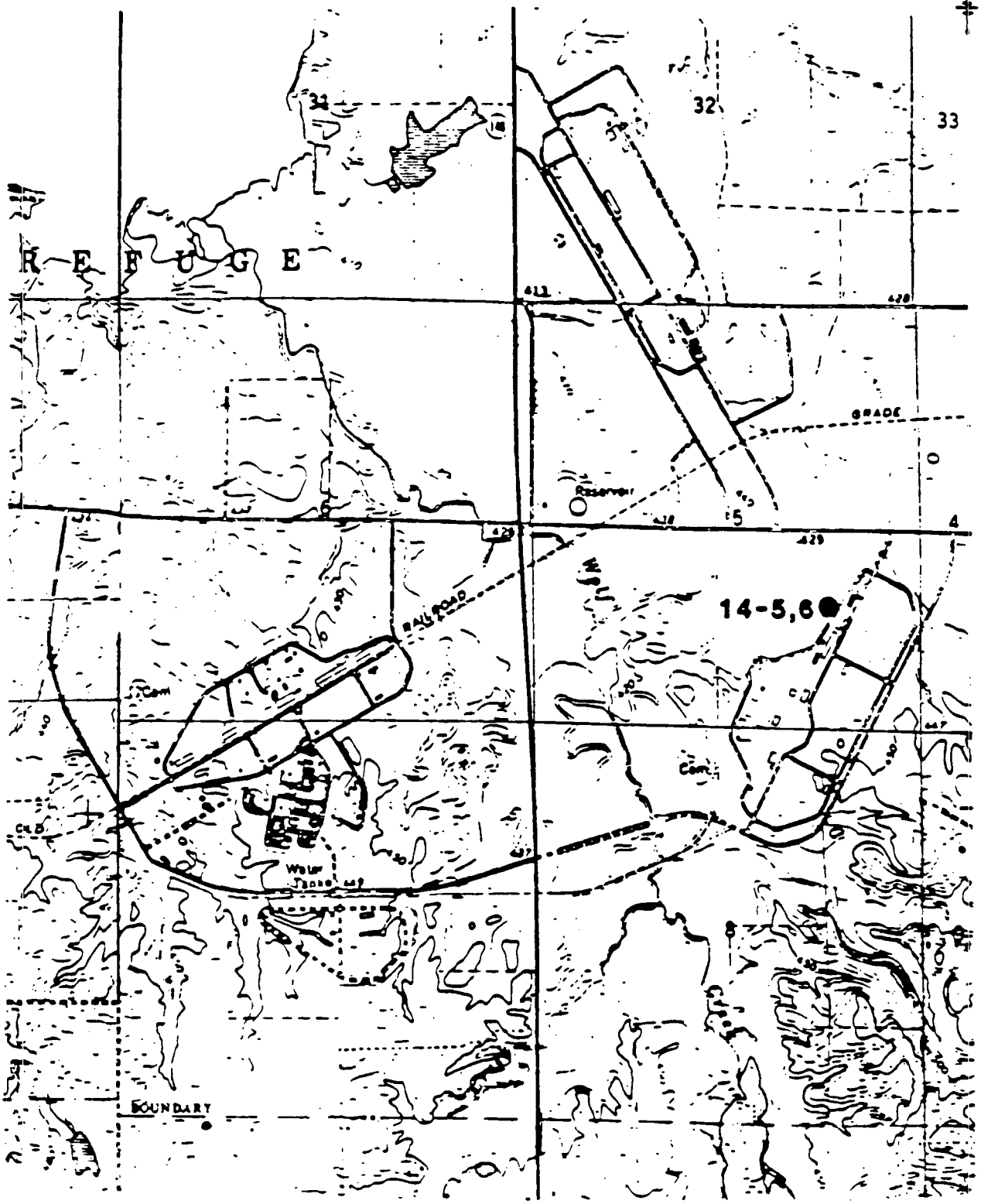

SITE 14
AREA 14 SOLVENT STORAGE
PHASE I
N
14-1
DITCH
14-2
DRUM STORAGE
STANDPIPE DISCHARGE
D
INDUSTRIAL BUILDING
14-3
CHEMICAL STORAGE TANKS
14-4
DRUM STORAGE
D - DECONTAMINATION AREA
APPROXIMATE SCALE
0
100
200
(FEET)

FIGURE 21-2

SITE 14
SAMPLING LOCATIONS
PHASE II



21.3 Analytical Results (See Appendix I, page 14)

21.3.1 Phase I Analytical Results

The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported below are thus estimated values and some compounds which were not detected may in fact be present.

Chloroform was detected in the water samples at 11 and 43 ug/L, which was above the ambient water quality criteria for human health of 0.19 ug/L. Iron and manganese levels in water (600 and 180 ug/L) were above the Federal MCLs and Illinois Public Water Supply Standards, but only the manganese concentration was above the Illinois General Use Water Supply Standard of 150 ug/L. Iron and manganese standards are based on concerns for taste and color, such that the levels detected are not indicative of any site risks. Bromodichloromethane was also detected at a concentration of 5.0 ug/L. Subsequent to an FID screening of 36,704 ug/kg, one of the sediment samples was selected for CLP organics. This sample contained trace organics below the detection limits or at levels consistent with those detected at the control sites or in the QA/QC blanks. N-nitrosodimethylamine (95 ug/kg wet weight), methylene chloride (657 ug/kg wet weight), and acetone (188 ug/kg wet weight) were among the compounds which were detected.

21.3.2 Phase II Analytical Results

Analytical results for the water sampled showed 123 ug/L chloroform, as well as traces of acetone (43 ug/L) and bromodichloromethane (23 ug/L). Methylene chloride was detected in the water (3-15 ug/L) but

also in the QA/QC blank. The spike recoveries for the CLP analysis in water were outside of QC limits. All parameters were below the Federal MCLs and Illinois State Public Water Supply Standards, but chloroform exceeded the AWQC for human health of 0.19 ug/L. The sediment contained 4-methylphenol (273 ug/kg), bis-(2-ethylhexyl)phthalate (270 ug/kg), di-n-butyl phthalate (1680 ug/kg), acetone (6480 ug/kg) and methylene chloride (676 ug/kg). No other organics were detected in the sediments; however the volatiles were analyzed outside the holding time and the semi-volatiles were outside the QC limits for spike recoveries. All other parameters were detected at concentrations similar to those detected at the control sites.

21.4 Environmental Effects

21.4.1 Qualitative Assessment

21.4.1.1 Source Evaluation

As discussed in the preceding sections, the Solvent Storage Site is a ditch which receives effluents from an active manufacturing facility. Its upgradient proximity to Crab Orchard Lake creates the potential for offsite contaminant transport to the lake. Several compounds were detected in ditch water and sediment including N-nitrosodimethylamine, methylene chloride, acetone, 4-methyl- phenol, phthalate esters, chloroform, and bromodichloromethane. Of the contaminants in water, only chloroform exceeded Federal water quality criteria.

The physicochemical and toxicological properties of site contaminants of concern are summarized in Exhibit A. The detection of N-nitrosodimethylamine in ditch sediments is significant due to concerns that it might be carcinogenic in humans. However, the residues detected

are of the same order of magnitude as detected at the control sites, and are not supported by QA/QC data. For these reasons, N-nitrosodimethylamine will not be considered further in this risk assessment. Di-n-butylphthalate was detected in sediments but will not be used as a site indicator parameter since it was also present as a laboratory contaminant. In addition, due to its small size and intermittent dry periods, the ditch does not apparently support significant population of aquatic organisms which might be exposed to phthalates or other site contaminants. The volatility of chloroform and bromodichloromethane compounds from water is high and it is highly unlikely that these residues would persist for a sufficiently long period of time to be transported offsite (Callahan et. al., 1979). For these reasons, chloroform and bromodichloromethane detected in water at this site were not chosen as site indicator compounds. In contrast, methylene chloride was detected in both water and sediment matrices and thus, although it may be highly volatile from water, its presence in sediments will result in its persistence in the stream. It should be noted that methylene chloride was detected at very low levels in the water column as well as in the water blanks; it was not detected in the QA/QC blank for sediment.

Due to persistence, relative residue levels, and concerns for toxicity, methylene chloride was chosen as the site indicator contaminant. Methylene chloride is of concern due to its ability to induce damage to several organs in animal studies, mutagenicity, and evidence suggesting that it may be carcinogenic in animals. Available data show a low degree of toxicity in aquatic organisms. Although methylene chloride may also be formed during the chlorination of plant effluents, it is also a major industrial solvent. Its presence in the ditch sediments at this site in

relatively high concentrations suggests that it may have been discharged as a waste solvent.

21.4.1.2 Transport Route Evaluation

The site indicator contaminants have been detected only in the sediments of the drainage ditch. In view of the high water solubility of methylene chloride, residues in water might be expected. However, methylene chloride is also very volatile from water and may evaporate immediately under site conditions after discharge or desorption from the sediment.

a) Air: Due to the high volatility of methylene chloride, the airborne exposure route is functional.

b) Direct Contact: Due to the presence of sediments containing site contaminants in the ditch, the direct contact pathway is considered functional.

c) Surface Water: Even though site indicator residues were not detected in the water column in significant levels, their presence in sediments creates a functional pathway for offsite transport via surface runoff.

21.4.1.3 Receptor Evaluation

Human

Given the industrial nature of the site and non-populated downgradient land, the human receptor population will be limited to facility employees, site intruders, and, potentially, occasional hikers in downgradient areas.

a) Air Route: Since methylene chloride residues may volatilize from the ditch sediments and water, it is possible that site employees or intruders in the immediate vicinity of the plant may be exposed to methylene chloride vapor.

b) Direct Contact: Given the size of the manufacturing facility and the probable volumes of aqueous discharge, standing water will probably almost always be present in the ditch. Therefore, the likelihood that human receptors will come into direct contact with contaminated sediments is virtually nil, and this route is judged to be incomplete for human exposure.

c) Surface Water Route: Due to the rapid volatility of the site contaminant from water, human exposure to site contaminants transported off site are not probable, and this route will not be considered further.

Wildlife:

The most likely wildlife receptors to be exposed to site indicator residues include terrestrial wildlife in the immediate area of the source discharge, and downstream aquatic organisms which may inhabit the drainage system. Due to the small size of the ditch, it is subject to dry periods and therefore is not likely to support a significant aquatic population.

a) Air Route: As with human exposure by this route, any terrestrial wildlife in the immediate vicinity of the plant discharge may be exposed to vapors of methylene chloride. The lack of exposed contaminated dusts will minimize inhalation exposures to dust-bound contaminants.

b) Direct Contact: Benthic and bottom-feeding aquatic organisms may be exposed to site contaminants in the sediments of the drainage system downstream if sediments are transported offsite by surface runoff.

c) Surface Water: Since site indicator contaminants were not found in the water of the ditch, wildlife exposures by this mechanism will not occur.

21.4.2 Quantitative Assessment

21.4.2.1 Estimates of Release and Exposure

a) Air Route: The qualitative assessment determined that the air exposure route was potentially complete for humans and wildlife in the immediate vicinity of the source discharge due to the volatility of methylene chloride detected in sediments. Methylene chloride residues detected in the water column of the ditch were low, approximately 15 ug/L. Without data on the total volume of contaminated water at this location, an estimation of the steady state air concentration of methylene chloride is not possible. It is most likely that volatilized material will be immediately diluted to non-detectable levels and carried offsite by wind.

b) Direct Contact: The qualitative assessment determined that the direct contact route was complete for benthic and bottom-feeding aquatic organisms living downgradient in the site drainage system. These organisms may inadvertently ingest site contaminant while feeding within the sediment. Data on this ingestion rate were not located. For the purpose of this assessment, it is assumed that a benthic organism (i.e. annelid, insect larva) ingests 1 mg sediment

per g body weight. This would result in a daily intake of 0.7 ug/kg methylene chloride; the significance of this exposure is discussed in the quantitative assessment.

c) Surface Water: The surface water exposure route is complete for aquatic organisms inhabiting the drainage ditch. Methylene chloride was detected at 15 ug/L, these site residues will dissipate rapidly from the water by evaporation, unless continuously replenished by plant discharges. The quantitative assessment will consider the potential risk from these residues.

21.4.2.2 Quantitative Risk Assessment

Humans:

The qualitative assessment determined that the only complete exposure route for humans at this site would be acute or sub-acute inhalation exposure of plant workers to methylene chloride at levels which in all likelihood are below the limit of analytical detection. The health effects of methylene chloride are produced by chronic exposures and the acute toxicity of this compound is quite low. Therefore, it is concluded that short-term, acute exposure to methylene chloride residues which may exist at this site pose a negligible human risk.

Wildlife:

As reasoned above, exposure of terrestrial wildlife to airborne residues of methylene chloride poses a negligible risk due to the low levels of contaminant present. Aquatic organisms dwelling in the ditch downgradient of the source discharge, if any, may be exposed to concentrations of methylene chloride on the order of 15 ug/L,

without accounting for dilution of this contaminant in downstream areas where the ditch is somewhat larger. This concentration poses no concern for acute toxicity, although data on the chronic effects of methylene chloride are lacking. Downstream dissipation of residues would rapidly reduce chronic risks, if any, from methylene chloride exposure.

21.4.3 Analysis of Uncertainties

The principal areas of uncertainty are whether there is the potential for a viable benthic community in the drainage system in the absence of potential toxicants, and the potential low level, long-term effects of methylene chloride.

21.5 Preliminary Remedial Alternatives

The contaminants detected in the site investigations included volatile and semi-volatile organics. Methylene chloride was present in both the water and sediments in the ditch, making this contaminant the most persistent. However, the levels in water were not considered to pose a risk to humans or aquatic organisms (if any may inhabit the ditch) in or in the vicinity of the ditch. Due to the detection of some volatile organics in water at levels above the AWQC, and the proximity of active manufacturing areas, periodic monitoring of ditch waters might be conducted. Attachment 1 details a Monitoring Program which might be implemented. The monitoring activities together with improved housekeeping practices in the area could provide adequate protection of the environment and humans at this site. No further evaluation of remedial alternatives will be considered.

21.6 Conclusions and Recommendations

It can be concluded that the Solvent Storage ditch does not pose a risk to potential animal or human receptors, although improved maintenance practices in the area might be observed. The sediments and water were found to contain methylene chloride but not at levels which would pose a threat to exposed populations. This site will not be considered further in the FS. A monitoring plan for volatiles and semivolatiles in water is included as Attachment 1.

SECTION 22 - SITE 15, AREA 7 PLATING POND

22.1 Site Description

Site 15 is a pond located within a wooded rise to the south of Area 7 (See Figures 22-1 and 23-1). The pond reportedly received plating wastewater from previous Olin operations at the site. An inlet pipe is located on the northern side of the pond. There does not appear to be any outlet. The plating pond has dimensions of approximately 50 feet x 30 feet, and is bermed to about 5 feet above the water level. The water in the pond was estimated to be about 4 feet deep at the time of the inspection. Frogs were observed in the pond.

22.2 Site Investigations

22.2.1 Phase I Site Investigations

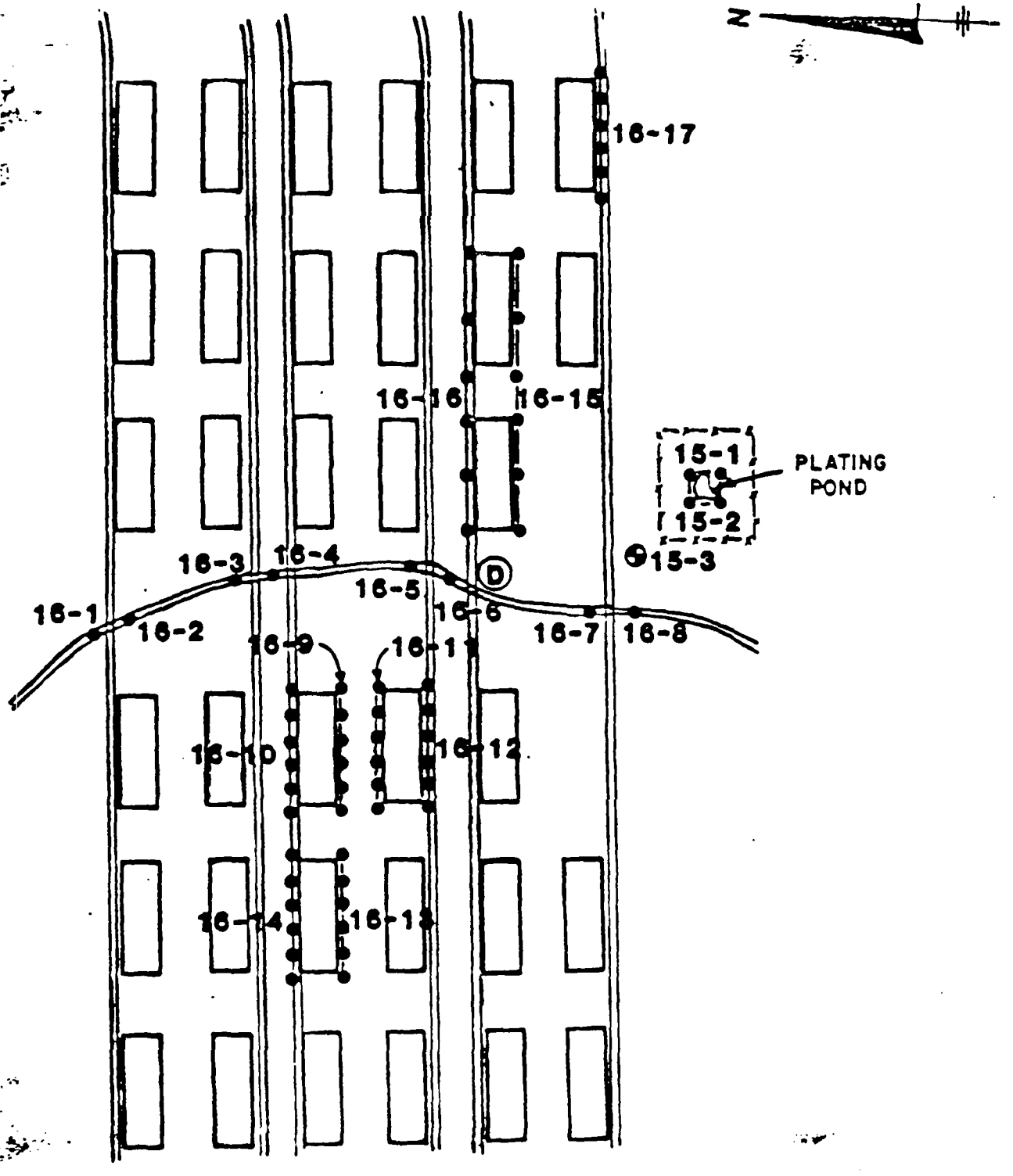
One composite surface water sample and one composite sediment sample (0-1 ft depth) were collected. The composites consisted of four grab samples, one from each corner of the pond. A downgradient monitoring well was installed to a depth of 15 feet in clayey silt and was screened over an interval of 5-15 feet.

22.2.2 Phase II Site Investigations

A ground water sample was collected from the monitoring well installed in Phase I. The purpose of this sampling was to determine if contaminants present in the pond have migrated to ground water. Due to an oversight, two piezometers which were scheduled for this site were not installed. One composite sample of pond sediment was collected for total and extractable chromium analysis. The purpose of this analysis was to

FIGURE 22-1

SITE 15-AREA 7 PLATING POND
SITE 16-AREA 7 INDUSTRIAL PARK
PHASE I



D - DECONTAMINATION AREA

APPROXIMATE SCALE

0 100 200
(FEET)

determine if the pond sediments represent a source of leachable chromium or exhibit the characteristic of a hazardous waste.

22.2.3 Site Hydrogeologic Characterization

22.2.3.1 Site Geology

Based on results of the test boring procedure, the subsurface unconsolidated overburden consists of a medium to dark brown clayey silt with some fine sand as was identified in Boring 15-4. This material is present from the ground surface to at least 15 ft. in depth (total depth of boring). Bedrock was not encountered in the boring and therefore depth to bedrock and bedrock lithology is not known. As only one monitoring well was installed, the lateral extent and variability of the overburden is also unknown.

22.2.3.2 Site Hydrogeology

Shallow ground water occurring beneath the site was found at a depth of 2 to 4.5 ft. below the ground surface within the clayey silt soil unit during June 1987. The monitoring well installed screened this upper water table. Ground water elevations collected during the winter and summer of 1987 (wet and dry seasons, respectively) indicate a water table fluctuation of 2.5 ft. with water levels dropping during the summer months (Table 4-3). Figure 34-4 illustrates the monitoring well location and the ground water elevation of 18 June, 1987.

22.3 Analytical Results (See Appendix I, Page 15)

22.3.1 Phase I Analytical Results:

The pond water analyses did not indicate the presence of contaminants at concentrations above Illinois General Use or Public Water

Supply Standards but iron (1000 ug/L) exceeded the Federal MCL of 300 ug/L. The MCL for iron was based on aesthetic concerns, however, and does not represent a risk to human health or the environment. The sediment contained 508 mg/kg chromium, but chromium was not detected in the water. The metals concentrations detected were estimated for screening purposes (Exhibit B). The phosphorus level was somewhat elevated (1621 mg/kg) in the sediment. The pesticide alpha-endosulfan was also detected in the sediment (811 ug/kg), but not in the pond water.

22.3.2 Phase II Analytical Results:

EP Toxicity chromium in the extract from the pond sediment was below the detection level of 0.05 mg/L, less than the 5.0 mg/L criteria for EP Toxicity (40 CFR 261.24). The sediment analysis for total chromium was not completed due to an oversight in laboratory scheduling. The ground water contained total and filtered chromium of 15 and less than 1 mg/L respectively. Filtered concentrations of arsenic, cadmium and lead were below detection limits in the ground water and thus were within the Federal and state standards. Total metals concentrations were also within the standards; total arsenic and lead were 7.6 and 22 ug/L, respectively, and total cadmium was below the detection level of 5 ug/L. No volatiles or pesticides were detected, but PCBs were detected at 0.01 ug/l which is above the AWQC for human health. All other concentrations in water were within the Federal and State drinking water standards.

22.4 Environmental Effects

22.4.1 Qualitative Assessment

This site was chosen for investigation based on its history of use for

receiving spent plating wastewater. Since there did not appear to be an outlet from the pond, the only viable transport mechanism for migration of contaminants off-site was ground water.

Phase I investigations detected traces of chromium, phosphorous and alpha-endosulfan in the sediment, with chromium and alpha-endosulfan exceeding the typical Refuge background levels. However, the pond water analysis did not indicate the presence of contaminants at concentrations above Illinois Public Water Supply Standards. Supported by Phase I results, the EP Toxicity test for chromium in the Phase II resampling of pond sediment, was below the detection level. The levels of chromium and cadmium in ground water were below Illinois Public Water Supply Standards, but unfiltered arsenic and lead levels were slightly higher than the standards. The filtered concentrations for arsenic and lead were below the detection limits. Trace PCB concentrations were detected in the ground water, but the results were questionable since PCBs were not detected in other samples from the site or from adjacent sites, and past land use activities in the area do not support their presence.

Human activities in this area are limited to occasional visits by authorized Refuge personnel and persons using the storage buildings in Area 7. The plating pond is located in an elevated area of the site, removed from the roadway, and restricted by dense vegetation and a barbed wire fence. Therefore, human exposures would be of a short-term, non-chronic nature. Under present conditions, it can be concluded that the site does not represent a significant risk of chemical exposure to either humans or wildlife; however, some remedial action may be warranted to permanently decommission the pond.

22.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

22.4.3 Analysis of Uncertainties

The information relied upon for evaluating Site 15 included a history of plating wastewater discharge to the pond, a site inspection and sample analyses.

Chemical residue information consisted of analytical results on one composite surface water sample and one composite sediment sample. Since contamination of the site occurred through the discharge of plating waste into a pond with no outlet, the area of contamination is likely to be limited to the water and the surrounding sediment.

It can be concluded that the data generated and the review of historical information are adequate for evaluation of the remedial alternatives for this site. The sampling analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

22.5 Preliminary Remedial Alternatives

The analytical results and risk evaluation presented in the previous sections indicated that this site does not contain contaminant levels that would be detrimental to the environment or to human or wildlife receptors. However, since the pond is no longer active, remedial measures for closure will be evaluated as part of the FS.

22.6 Conclusions and Recommendations

It can be concluded that although the plating pond site does not currently represent a significant risk of exposure at the Refuge, further evaluation is recommended as part of the FS to determine options for permanent closure of the pond.

SECTION 23 - SITE 16, AREA 7 INDUSTRIAL SITE

23.1 Site Description

Area 7 consists of a complex of 33 identical buildings over an area of 55 acres which have been used for a variety of industrial purposes during the past forty years. The buildings are arranged in six rows, each of which were at one time served by a railroad siding. Most of the buildings are currently used for dry warehousing purposes. However, three buildings are used by Pennzoil for waste oil recovery and recycling operations. Black residues are noticeable around some of these buildings. Two other buildings are used by a refurbisher of mining equipment. Black residues are also evident around these buildings. A drainage channel runs from south to north through the site.

Site 16 consists of the area in the vicinity of the five identified buildings and the drainage ditch (see Figures 22-1 and 23-1).

23.2 Site Investigations

23.2.1 Phase I Site Investigations:

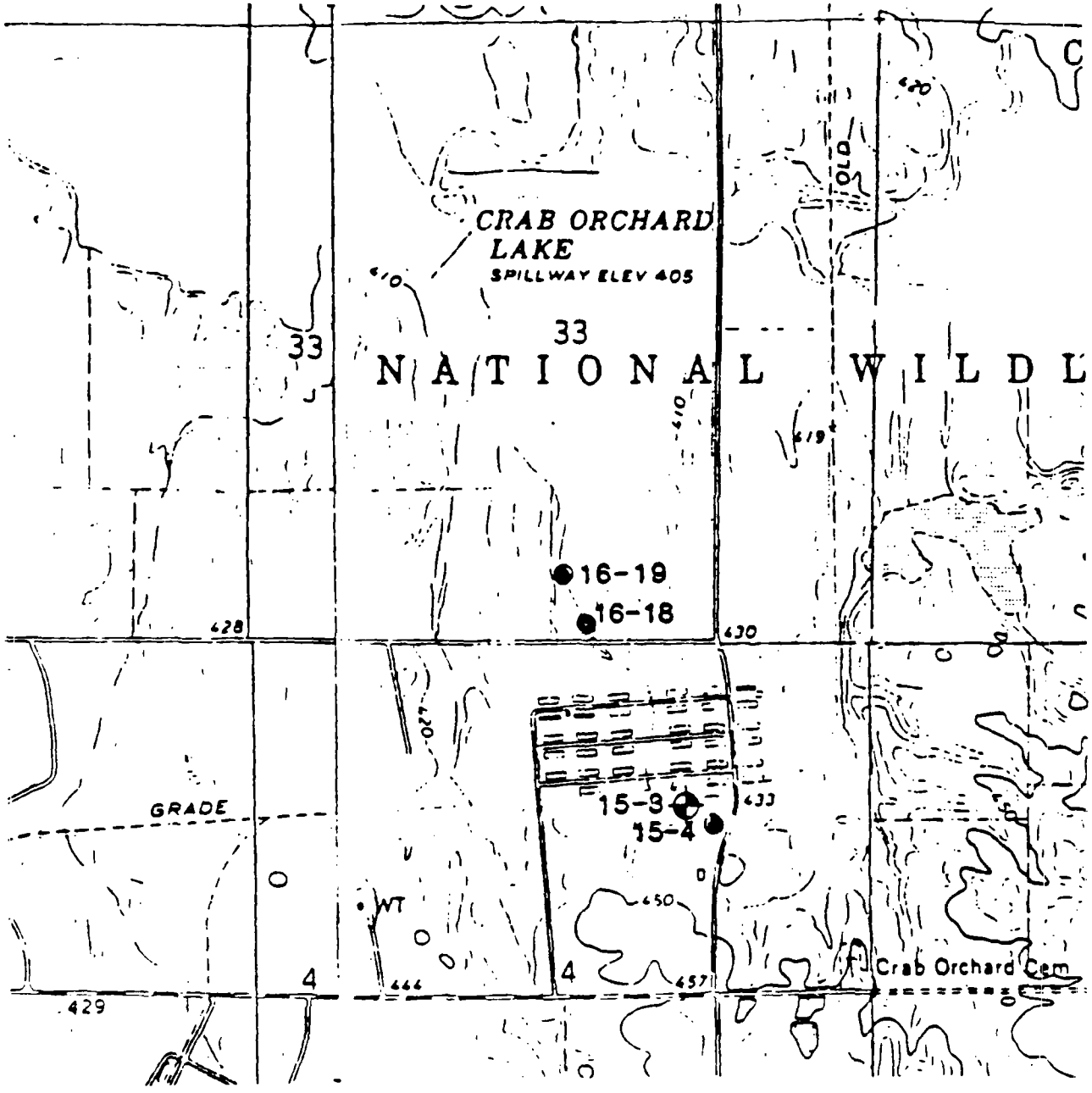
Two composite surface water samples and three composite sediment samples (0-1 ft depth) were collected from the drainage ditch. Nine composite soil samples (0-1 ft depth) were also collected from those areas which exhibited black oily soil residues in the vicinity of several manufacturing buildings. Three samples (one sediment and two soils) were resampled for full CLP organics analyses.

23.2.2 Phase II Site Investigations:

To define the areal exter of contaminant migration, one composite surface water and one composite sediment sample were collected from the

FIGURE 23-1

SITES 15 & 16
SAMPLING LOCATIONS
PHASE II



N

◆ Shallow well

SCALE IN FEET

0 1400 2800

drainage ditch at locations downstream from the Phase I locations (See Figure 23-1). The Phase II samples were analyzed for magnesium, lead, arsenic, purgeables, pesticide/ PCBs, and base/neutrals/acid extractables. The purpose of these analyses was to determine if surface residuals from the Area 7 buildings were migrating along the ditch toward Crab Orchard Lake.

23.3 Analytical Results (See Appendix I, Page 16)

23.3.1 Phase I Analytical Results:

One of the two water samples from the drainage channel contained low concentrations of organics including chloroform (77 ug/L) and carbon tetrachloride (66 ug/L). The detected volatile compounds were above the AWQC for human health of 0.19 ug/L for chloroform and 0.4 ug/L for carbon tetrachloride but not above the correspondir criteria for aquatic life protection. Both samples contained magnesium (25 and 36 mg/L) and manganese (340 and 70 ug/L). No parameters were at concentrations above Illinois General Use Water Standards, but manganese levels were above the Illinois Public Water Supply Standards and the Federal Drinking Water MCLs. Manganese levels are not present at concentrations which would pose any concerns for public health. Two soils and one sediment selected for CLP organics contained trace organics (estimated concentrations below the detection limit or, in the case of acetone, detected in the QA/QC blank). The organics detected were, on a wet weight basis, acetone (348-782 ug/kg), N-nitrosodimethylamine (not detected to 115 ug/kg), crysene (not detected to 453 ug/kg), pyrene (not detected to 356 ug/kg), fluoranthene (not detected to 389 ug/kg), and low microgram levels of other base/neutral compounds,

possibly due to the oil spillage indicated by the black residues. Most of the organics detected were estimated values since the reported concentrations were below the analytical detection limits. In addition, the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present. Metals concentrations are estimated. All other concentrations in soil samples were consistent with those detected at the control sites.

23.3.2 Phase II Analytical Results:

Acetone and methylene chloride were detected in the water sample but at levels attributed to blank contamination. Aldrin (0.17 ug/L) and dieldrin (0.54 ug/L) were present at concentrations below the Illinois Public Water Supply Standards but above the AWQC for humans and for aquatic life. Holding times were exceeded for the volatiles analysis, and the recoveries for duplicate and spike samples were outside the QC limits for the volatiles and pesticide analyses. The dissolved magnesium concentration was 26,300 ug/L.

Traces of volatile compounds, including chloroform and chlorobenzene, were found in the sediment, as well as 1600 ug/kg di-n-butyl phthalate. The latter was also detected in the QA/QC method blank sample. No other organics were detected. Magnesium (958 mg/kg), lead (13 mg/kg) and arsenic (10 mg/kg) were detected in the sediment sample.

23.4 Environmental Effects

23.4.1 Qualitative Assessment

This site was chosen for investigation based on its past and current use as a site for storage activities, waste oil recovery, and recycling operations. Observable black residues, possibly originating from oil spills, suggested that this site might be a source of waste material. The drainage ditch provides a viable mechanism for transport of potential contaminants.

Phase I investigations detected chloroform and carbon tetrachloride in one water sample from the ditch. Both water samples contained magnesium but not at concentrations considered to threaten wildlife or affect human health. The soils and sediment samples from Phase I did not contain any contaminant concentrations that were significantly above Refuge background levels. Phase II data support the Phase I results; traces of aldrin and dieldrin were detected in the water sample but were below Illinois Public Water Supply Standards and Federal drinking water standards although above the more stringent AWQC for these parameters. Magnesium was again detected in the water but not at levels that would represent a concern. Magnesium, lead, and arsenic levels in sediments were similar to the concentrations found at the control sites. Di-n-butylphthalate was detected but was also present as a contaminant in the QA/QC blanks. N-nitrosodimethylamine was detected in Phase I but at levels similar to those at the munitions control site, or at approximately one half the levels determined to be acceptable for potentially exposed human or wildlife receptors at Site 17, Section 24.4.

The sampling results from both Phase I (upstream) and Phase II (downstream) did not indicate any evidence of migration of contaminants off-site. Since trace contaminants were detected, the presence of a waste

source may be justifiable; however, the small intermittent ditch does not support significant aquatic life, and human visitors to the site are very rare because the area is used for long-term storage and not for day-to-day industrial operations or residential purposes. Potential human exposures at the site might consist of occasional visits for 2-3 hours for purposes of loading or unloading supplies to storage. Since no receptors at the site could be exposed to chronic risk levels, and the levels of contaminants do not represent a risk for acute (short-term) exposure, it is not possible to have a "complete" exposure scenario. Therefore, on the basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

23.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

23.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a site inspection and the sample analyses. Black residues were noticeable around the buildings on the site, suggesting that they may be a source of contamination.

Chemical residue information consisted of analytical results on surface water, sediment and soil samples. This information was obtained only for the top one foot of soil; deeper soil borings and ground water monitoring were not conducted. Since any contamination of the site would

have occurred through a variety of spills and leaks during storage and loading rather than through excavation and burial, the area of contamination would be limited to the surface soil and sediment. The samples collected should adequately represent the conditions of the site.

It can be concluded that the data generated are adequate for evaluation of the remedial alternatives for this site. The sampling analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

23.5 Preliminary Remedial Alternatives

The analytical results discussed in the previous sections indicate that site constituents have not migrated and thus do not represent a risk for offsite transport of contaminants. The low levels of constituents found do not represent a threat of exposure to humans, aquatic life, or terrestrial wildlife. However, due to intermittent storage activities in this area, it is recommended that a follow-up monitoring program be conducted. Attachment 1 details the frequency and sampling plan recommended. The parameters of interest for monitoring in water include volatile organics (chloroform, carbon tetrachloride), semivolatiles, and pesticides (aldrin, dieldrin).

23.6 Conclusions and Recommendations

It can be concluded that the Area 7 Industrial Site does not pose a risk to humans or wildlife. No further evaluation in the FS is recommended, however, a monitoring program for volatiles, semivolatiles and pesticides in water (Attachment 1) might be implemented.

SECTION 24 - SITE 17, JOB CORPS LANDFILL

24.1 Site Description

Northeast of the Refuge Water Works is a ten-acre pond created by the Job Corps in the mid-1960s (See Figure 24-1). Attention was brought to this pond in early 1985 because thirty or more geese carcasses were found floating on the water or littering the shores. Some of these carcasses were relatively fresh while others were in various states of decay. The Fish and Wildlife Service has completed extensive analyses of these carcasses and has ruled out a variety of potential chemical causes although a definite conclusion was never reached. The "Job Corps Landfill" was discovered while investigating the geese kills.

Site 17 is comprised of the pond and adjacent one-acre landfill located north of the pond in a wooded area. The area contains widespread debris, such as bottles and cans, which do not appear to be buried deeply. Mica flakes, small electrical contacts, and a few small capacitors have been observed among the debris.

Aerial photos are available for the area near the Job Corps Site for 1951, 1960, 1965, 1971, and 1980. This sequence provides information on the usage of the Job Corps area. The 1951 photo shows a clearly defined fan-shaped dumping area with an access road entering from the southeast. This access road suggests that dumping in this area was conducted over an extended period. The access road is a well-defined one-lane path.

The 1960 photo indicates that the site was inactive at this time. The landfill area shows some vegetative cover and the contrast with surrounding vegetation is less distinct than in the earlier photos. The access road is still visible, but covered by brush in areas, suggesting that it had not been used

FIGURE 24-1

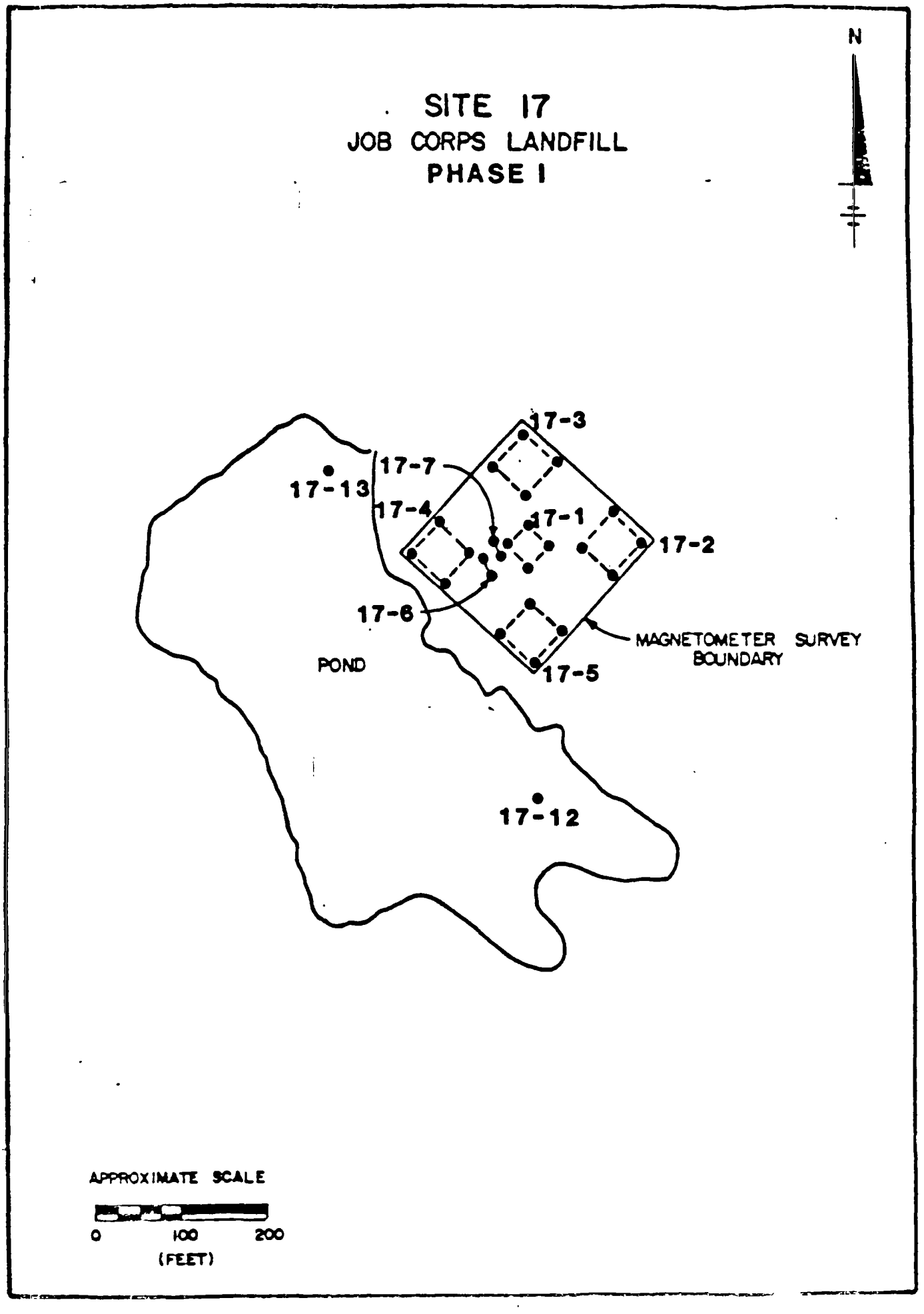
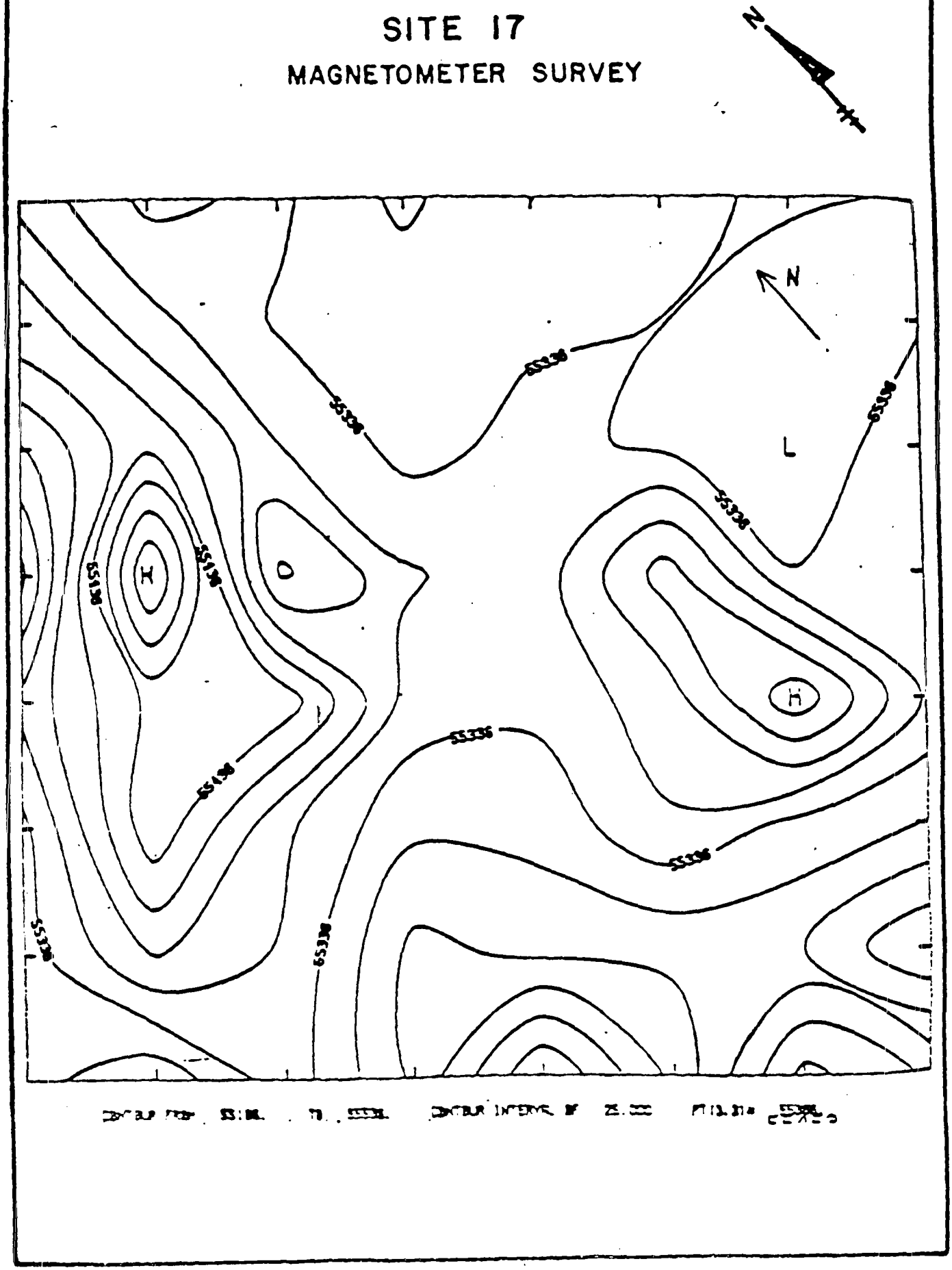

N
SITE 17
JOB CORPS LANDFILL
PHASE I
17-3
17-7
17-13
17-4
17-1
17-2
17-6
MAGNETOMETER SURVEY BOUNDARY
POND
17-5
17-12
APPROXIMATE SCALE
0
100
200
(FEET)

FIGURE 24-2

SITE 17

MAGNETOMETER SURVEY



for some time. The 1951-1960 timeframe is consistent with a 1957 automobile license plate observed during the site inspection.

In the 1965 photograph the access road is distinguishable only as a boundary of an adjacent planted field. The landfill area is not covered by brush or trees, and the vegetative cover appears similar to that in the surrounding area.

The 1971 photo shows the Job Corps pond which was created in the mid-1960s. The eastern end of the pond overlies an area which had been the access road. The landfill area shows signs of trees, brush and larger vegetation. The 1980 photo shows the site much as it is today, with trees and heavy vegetation overlying the fill area.

During the site inspection, various articles typically associated with municipal refuse were noted, including bottles, plates, etc. The site is presently covered with dense vegetation and the access road is no longer useable.

24.2 Site Investigations

24.2.1 Phase I Site Investigations:

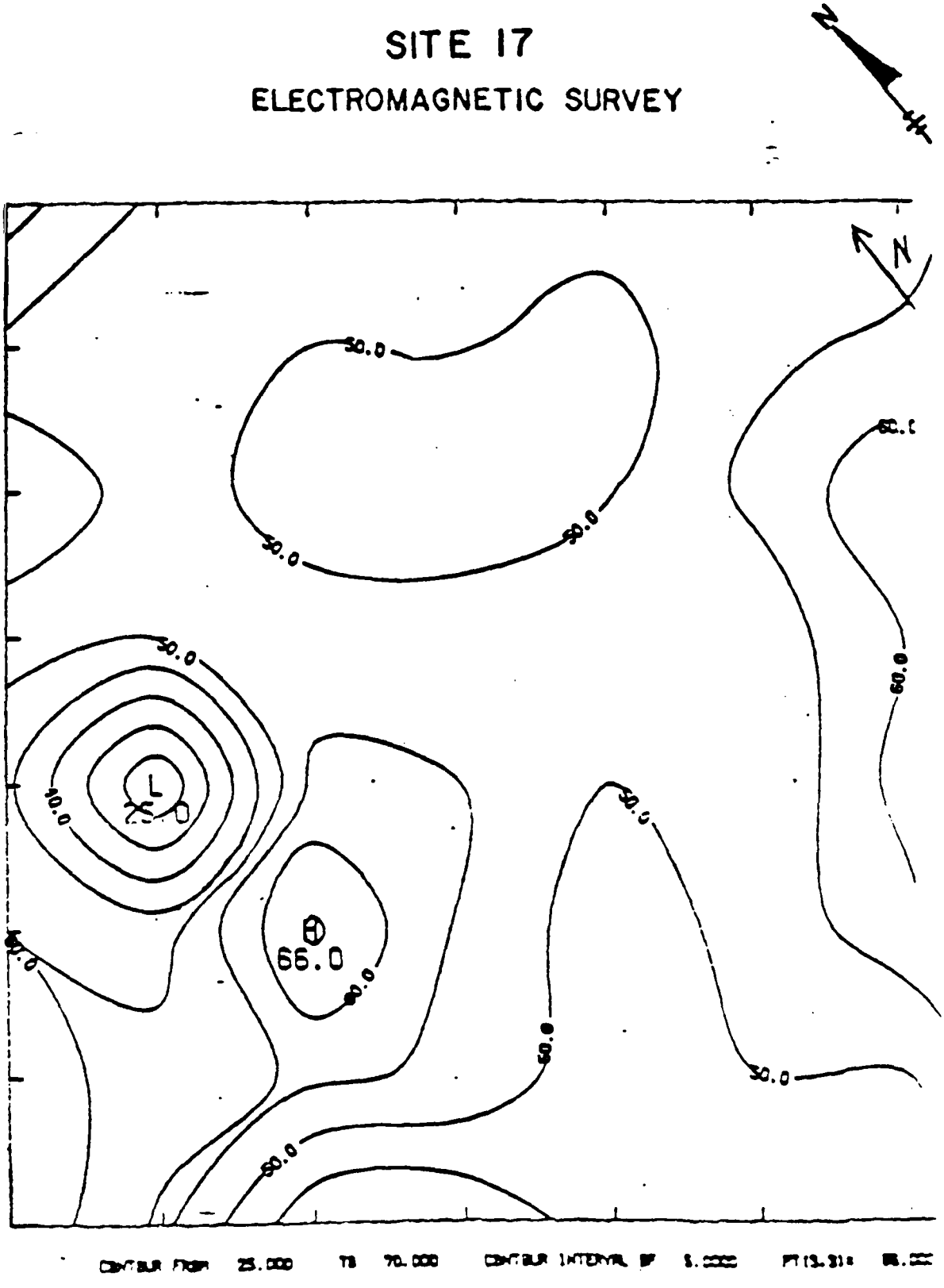
A magnetometer and electromagnetic terrain conductivity survey was conducted. The results of these surveys are shown on Figures 24-2 and 24-3.

Two composite surface water samples were collected from the pond, and seven composite soil samples (0-1 ft depth) were collected from the landfill. A composite was collected from each of five 50 ft x 50 ft squares within the landfill as well as from two bare patches. Figure 24-1 depicts the Phase I sampling locations.

FIGURE 24-3

SITE 17

ELECTROMAGNETIC SURVEY



24.2.2 Phase II Site Investigations:

Phase II studies were intended to better define the vertical and horizontal extent of the contamination identified in Phase I (See Figure 24-4). Forty-seven soil samples were collected at the landfill, thirty-five at depths of 0-1 ft and twelve to a depth of 3 ft. All soils were analyzed for PCBs, lead and cadmium. In addition, twelve of these samples were also analyzed for explosives and nitrosamines. One Phase I soil (location 17-1) was reanalyzed for mercury. Two composite surface water and six grab sediment samples were collected from the pond and analyzed for PCBs, lead, cadmium and explosives. The pond water samples were also analyzed for CLP organics, PCBs, arsenic, and copper. Three surface (0-1 ft. depth) sediment samples were collected along the shallow embankment and three were collected from the deeper area of the pond.

Five monitoring wells, four shallow and one deep, were installed to depths of 12 to 30 feet (see Figure 24-4). The wells were set in silt and silty clay soils, except well 17-65 which was set on top of rock. Each well utilized five-foot stainless steel screens. Ground water samples were analyzed for CLP HSL organics and metals, low-level PCBs, nitrosamines and explosives.

Two composite fish (one bass and one bluegill) were sampled from the Job Corps Pond shortly after the Phase II investigations. The samples were analyzed for pesticides, PCBs, cadmium, mercury and lead.

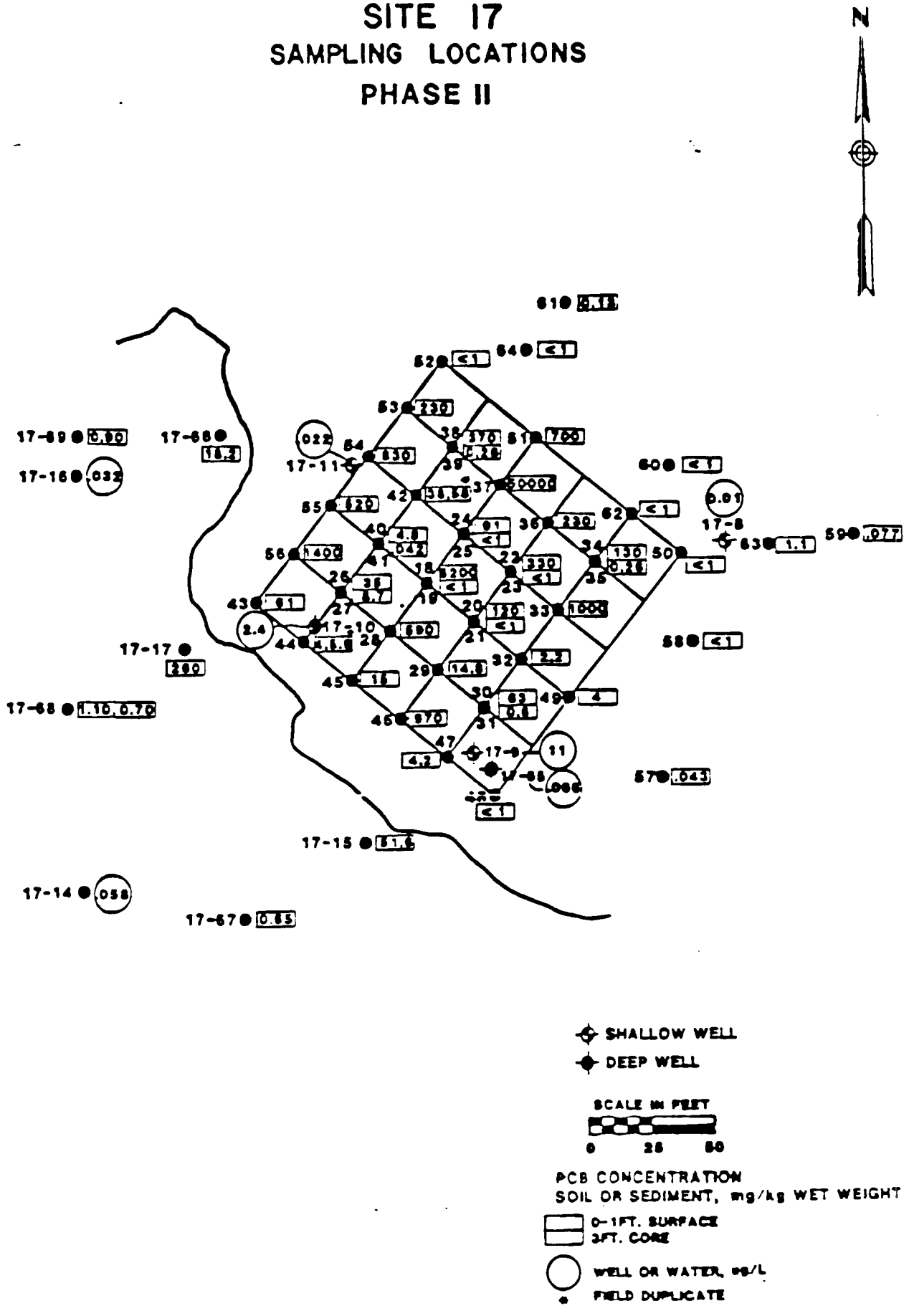
24.2.3 Site Hydrogeologic Characterization

24.2.3.1 Site Geology

The sequence of unconsolidated soils underlying the site to the maximum penetrated depth of 28 feet (Boring 17-65) consist of predominantly silty clay. Occasional organic silt layers with a trace of

FIGURE 24-4

SITE 17
SAMPLING LOCATIONS
PHASE II



fine gravel were identified in upper portions of Borings 17-9 and 17-10 and near a small swamp south of the site.

Underlying the silt and clay sequence in Boring 17-65, a shale bedrock was encountered that was gray and friable. The depth to bedrock was not identified in other borings.

24.2.3.2 Site Hydrogeology

Occurrence of Ground Water

Ground water was identified to occur about 6 to 8 feet below ground surface during June 1987 in the shallow wells. Ground water levels were about 5 feet higher during December 1986. The ground water occurrence in the deeper well set on top of bedrock exhibited a slightly lower hydraulic head.

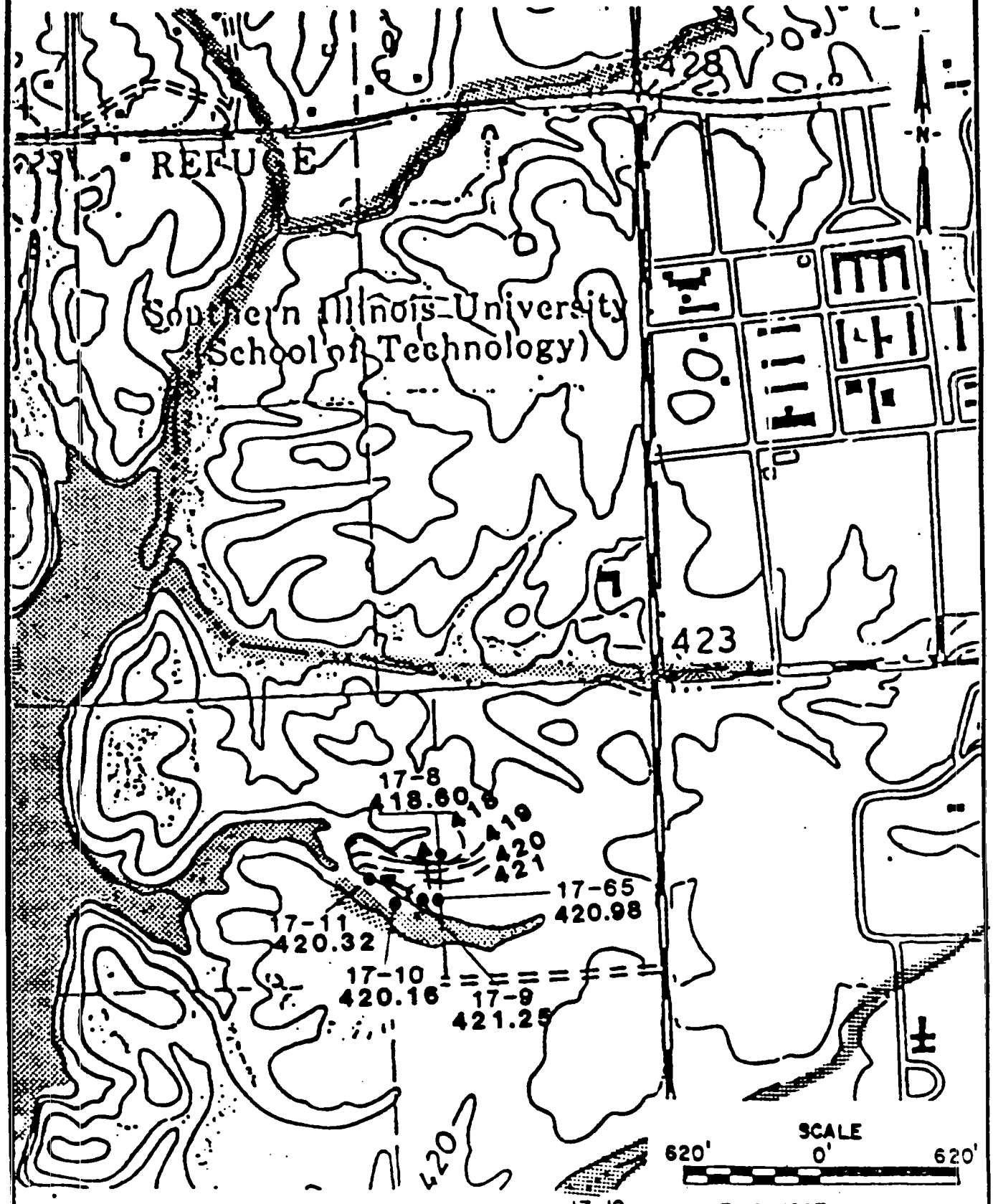
Ground Water Flow Conditions

Elevations of ground water within the water table wells were contoured and are shown on Figure 24-5. Results of these data taken from June 1987 indicate a radial groundwater flow pattern to occur in north, west, and south directions toward adjacent surface water drainage swales. Discharge from the swales which contain intermittent surface flows is toward Crab Orchard Lake about 1,000 feet west.

Ground water velocity calculations employing the formula given in Section 4.2 were performed utilizing an average hydraulic gradient of 0.2 ft/ft (June 1987), average hydraulic conductivity (K) of 9.36 ft/day, and a porosity of 0.35 (Davis, 1966). Resultant ground water flow velocity was calculated to be about 9.36 ft/day or 3,400 ft/year.

FIGURE 24-5

SITE 17 GROUNDWATER FLOW MAP



Vertical flow potential from the upper water bearing zone was found to be slightly downward into lower unconsolidated aquifers based on ground water elevations taken from well nest 17-65 17-9.

24.3 Analytical Results (See Appendix I, Page 17)

24.3.1 Phase I Analytical Results:

The magnetometer and electromagnetic survey shown in Figures 24-2 and 24-3 of the landfill did not reveal the presence of any concentrated pockets of conductive materials.

The surface soil samples collected from depths of 0-1 ft. from the landfill showed the widespread presence of PCBs and lead. All surface soil samples contained PCBs (Aroclor 1254), with concentrations ranging from 21 to 1700 mg/kg wet weight (28 to 2059 mg/kg dry weight). Lead was also present in all surface soil samples at concentrations of 609 to 14100 mg/kg wet weight. The detected PCB and lead levels at this site are not typical of the Refuge background. One soil sample contained 34 mg/kg cadmium and one soil contained 3 ug/kg mercury. All other metals were within the range of concentrations detected at the control sites. The metals concentrations performed in Phase I are estimated for screening purposes.

The explosive tetryl was detected in four soil samples at concentrations ranging from 3.7 to 6.5 mg/kg. One of the soil samples (location 17-2) analyzed for CLP organics contained on a wet weight basis, di-n-octyl phthalate (4760 ug/kg), 1984 ug/kg methylene chloride, and 173 ug/kg acetone. The FID scan of the same samples showed 127,895 ug/kg. Methylene chloride and acetone parameters were also present in the method blank. The second soil sample (17-6) analyzed for CLP organics (with 654,308 ug/kg FID scan) contained 220 ug/kg wet weight

N-nitrosodimethylamine, but all other organics detected were below the detection limit. One water sample contained 2.6 ug/L of 2, 4-DNT explosive residue, which is above the human health AWQC of 0.11 ug/L but below the aquatic life 24-hr average criterion of 230 ug/L. No explosives were detected in the other water sample or in the sediments or soils. PCBs and all other organics were undetected (below 10 ug/L) in the pond water samples, and all analyzed compounds were within the Illinois Public Water Supply Standards. It should be noted that the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data. (See Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present.

24.3.2 Phase II Analytical Results

PCB concentrations in surface (0-1 ft. depth) soils ranged from 0.077 to 50,000 mg/kg wet weight (0.101 to 69,042 mg/kg dry weight). Soil cores collected at 3 ft. depths, however, showed PCB concentrations below 1 mg/kg wet weight for all except one core (17-27) which contained 8.7 mg/kg wet weight (11.6 mg/kg dry weight). Figure 24-4 shows the areal distribution of PCB concentrations throughout the site on a wet weight basis. Lead concentration in soils ranged from below 6 to 174 mg/kg (dry weight) in the 0-1 ft. samples, and from below 6 to 13 mg/kg (dry weight) at 3 ft. depth with the exception of soil 17-27 which had 219 mg/kg lead. The highest lead concentrations were found to coincide with the highest PCB levels. Cadmium concentrations were relatively low and heterogenous. At the 0-1 ft strata, eight soil samples showed less than 1 mg/kg cadmium, while nineteen samples showed cadmium from 1 to

57 mg/kg, with a mean of 16 mg/kg. Cadmium concentrations were less than the detection limit of 1 mg/kg at the 3 ft. depth soil samples. The soil sample from Phase I (location 17-1) that was reanalyzed for mercury contained 190 ug/kg. No explosive residues were found in the soils. The pond bottom surface sediments contained between 0.65 and 260 mg/kg wet weight (3 and 607 mg/kg dry weight) PCBs, 4-49 mg/kg cadmium, and 38 to 5722 mg/kg lead. No other parameters were detected.

The groundwater and pond water samples contained traces of di-n-butyl-phthalate (2-12 ug/L, below the AWQC for human health of 35,000 ug/L), isophorone (40-115 ug/L, below the AWQC of 5,200-117,000 ug/L), acetone (22-4,270 ug/L), and methylene chloride (2-56 ug/L). Acetone, isophorone, and methylene chloride were also detected in the method blanks. The organics analyses were outside the QC limits for duplicate recoveries for some compounds were extracted outside of holding time.

The explosive nitrobenzene was detected in the water samples but not in the soils or pond sediments. Nitrobenzene concentrations (unfiltered) were 1.48 to 11.4 ug/L in the wells and 1.08-1.68 ug/L in the pond water, well below the AWQC of 19,800 ug/L (human health level), or 27,000 ug/L (aquatic life acute maximum).

Cadmium was below the 5 ug/L detection level (total and dissolved) for all pond and groundwater samples. Arsenic ranged from 7 to 27 ug/L total, and from less than 2 to the detection limit of 25 ug/L dissolved. Chromium ranged from 18 to 139 ug/L total and 1.7 to 6.4 ug/L dissolved in the wells, exceeding the standard of 50 ug/L for Illinois public waters and Federal drinking waters; chromium was not analyzed in the pond

water samples. Lead was detected between 4.3 to 55 ug/L total and 3.8 to 21 ug/L dissolved in groundwater, and between 1.6-3.6 ug/L total and less than 1 ug/L dissolved in the pond. Only one sample, ground water from well 17-11, exceeded the Illinois and Federal standard of 50 ug/L for lead. The ratios found between total and dissolved metals values indicated that the higher concentrations detected are mostly associated with suspended solids. All dissolved metals concentrations were below the Illinois Public Water Supply standards and Federal MCL standards.

PCBs (total aroclors) were detected in the groundwater samples at 0.01 to 11 ug/L. The higher range detected may have been due to suspended matter in the sample since most of the detected concentrations are higher than the solubility limit of PCBs in water. The pond waters contained 0.032 and 0.058 ug/L PCBs. All detected PCB concentrations were above the AWQC for human health and aquatic life protection. One well (17-65) contained traces of chloroform (12.5 ug/L), another well (17-10) had pentachlorophenol (19 ug/L), and bis(2-ethylhexyl) phthalate (23 ug/L). Well 17-9 also contained 10 ug/L of bis(2-ethylhexyl) phthalate. Chloroform levels exceeded the AWQC for human health of 0.19 ug/L; pentachlorophenol was above the criteria for protection of aquatic life (3.2 ug/L average), but not above the level for protection of human health (1,010 ug/L). The concentrations of phthalates were below the applicable AWQC criteria of 15,000 ug/L (human health). Two fish (a bass and a bluegill) sampled from Job Corps Pond were analyzed for pesticides, PCBs, and metals. No pesticides or PCBs were detected, but low levels of cadmium (0.05 mg/kg in bluegill only), lead (0.57 mg/kg in bass, 6.94 mg/kg in bluegill), and mercury (0.243 mg/kg in bass, 0.072 mg/kg in bluegill) were found.

Two composite fish (one bass and one bluegill) sampled from the Job Corps Pond were analyzed for pesticides, PCBs, cadmium, mercury and lead. No pesticides or PCBs were detected (detection limits 20 to 400 ug/kg). Low levels of cadmium (0.05 mg/kg in bluegill only) and mercury (0.243 mg/kg in bass, 0.072 mg/kg in bluegill) were detected. Lead concentrations were higher for bluegill, at 6.94 mg/kg, while only 0.57 mg/kg were detected in bass.

24.4 Environmental Effects - Job Corps Landfill

24.4.1 Qualitative Assessment

24.4.1.1 Source Evaluation

As described in Section 24.1, the site was likely to have been an area where mixed municipal and industrial type wastes were disposed. Field inspections of the site revealed various cans, bottles, containers and other articles typical of a mixed sanitary waste. However, the site inspection also noted the existence of a small exposed area containing wire electrodes and electrical components, suggesting that this location may have also received wastes of industrial origin. Geophysical investigations suggested widely-scattered debris rather than concentrated areas of fill material. The soil boring program encountered materials just below the surface (0-1 ft) consistent with what was observed at the surface.

Results of the soil and sediment sampling and analysis program indicate that the primary chemical compounds of concern at this location are PCBs, cadmium, and lead. Levels of cadmium at the site are very heterogenous making it difficult to derive accurate value for exposure in a quantitative risk assessment. In view of concern for cadmium toxicity to wildlife a qualitative risk assessment was completed. The

concentrations of PCBs detected, ranging from 0.077 to 50,000 mg/kg wet weight (0.10 to 69,042 mg/kg dry weight) are unusually high and inconsistent with what might be expected to be present within a landfill containing municipal wastes. Of the chemical compounds included and consistently detected at elevated levels in the sampling and analysis program, PCBs and lead are the most toxic. Therefore, for purposes of the quantitative risk assessment, it will be assumed that PCBs and lead are the compounds representing the highest risk. Since N-nitrosodimethylamine was also detected in a soil sample collected during the Phase I investigation, it will also be considered in the quantitative risk assessment for this site. Furthermore, for purposes of the nitrosamines assessment, and in order to provide a worst case conservative outcome in light of the uncertainty associated with the analytical result for this specific class of compounds, the concentration of nitrosamines in soil will be considered to be twice the level found in the single soil sample analyzed (i.e. 440 ug/kg rather than the detected level of 220 ug/kg). PCBs, lead and N-nitrosodimethylamine compounds are therefore used to define the waste "source". Exposure to cadmium in soil and sediment will qualitatively follow the exposure scenarios of PCBs and lead.

The physiochemical and toxicological properties of PCBs, nitrosamines and lead, as well as cadmium are summarized in Exhibit A. Lifetime dietary exposures of rodents to PCBs have established the carcinogenic potential of these compounds. PCBs produce a number of other chronic and subchronic effects as well. Lead exposure also presents cause for concern due to neurological, hematological and other effects demonstrated in humans and animals produced by chronic exposure to low lead levels.

There is insufficient evidence to determine if lead exposure presents a risk of cancer. Although dietary lead acetate has produced cancer in test animals, lead acetate would not be expected at the site due to chemical conversion to insoluble oxide and sulfate salts (USEPA ECAO-CIN-H055, 1984). PCBs were detected in surface soils in the landfill, as well as in the sediments of the adjacent pond. Based on the physiochemical properties of PCBs and lead, it can be assumed that they are strongly adsorbed to the surfaces of soil materials. They will therefore not be leached by runoff or surface infiltration, but will behave and be transported along with the particles to which they are adsorbed. There were no free liquid PCBs encountered during the investigation.

Because of the identification of PCBs in all of the soil samples collected, it is assumed that all of the samples within the defined boundaries of the site contain PCBs at levels in the range detected. For purposes of the quantitative risk assessment, the average value of 7,950 mg/kg dry weight PCB as Aroclor 1254 will be used as the worst case mean concentration, and the maximum concentration detected (69,042 mg/kg dry weight Aroclor 1254) will be used to represent the "upper bound" worst case residue. For lead, a soil level of 5,000 mg/kg dry weight was selected as a representative upper bound value for the site.

24.4.1.2 Transport Route Evaluation

a) Air: PCBs in soil exert a relatively low vapor pressure, but under certain conditions, transport of PCBs in the vapor state could constitute a functional transport route. Lead does not exert an appreciable vapor pressure, and therefore cannot be transported in the vapor state in significant quantities. However, because of the

existence of exposed, soil-adsorbed PCBs and lead at the site, dusts generated by wind erosion, vehicular traffic or activities by endogenous wildlife are likely to contain contaminants which can subsequently be transported by the air route. Therefore, it can be concluded that the air transport route can function to carry PCBs and lead and other waste related compounds to on- and off-site locations for subsequent exposures by receptors in those areas.

b) Direct Contact: Because of the existence of PCBs and lead in exposed wastes, soils, vegetation and sediments in the area, exposures by the direct contact route are possible.

c) Surface Water: There is a pond located directly adjacent to the waste area. The pond discharges to an adjacent stream through an engineered control weir, also adjacent to the waste area. Inspection of the site revealed the presence of waste materials at the shoreline, as well as submerged in the pond. The surface of the wastes in the site are approximately zero to four feet above the level of the pond. Therefore, it is likely that runoff from the waste site is directed into the adjacent pond.

In addition to the proximity of the exposed waste materials to surface water, the results of the sampling and analysis program indicate that the pond sediments contain PCBs. It is presumed that the PCBs in the pond sediments originated from the materials deposited in the landfill. On this basis,it can be concluded that the surface water transport route can function to transport waste materials from within the site to the pond, as well as by stream flow to offsite locations.

d) Ground Water: Results of analyses of the ground water indicate only minor residues of PCBs ranging from 0.01 ug/L to 11 ug/L. Lead was found at 4.3 mg/L to 55 mg/L. However, it is likely that the contaminants detected in the ground water samples are the result of the installation of the monitoring wells within a fill area containing PCBs and lead, rather than PCBs and lead which have migrated with the movement of water through the active mass. This behavior is consistent with what is known and previously observed regarding the movement of PCBs and lead in ground water and the absence of these constituents in subsurface soil. They are not expected to move with ground water to any appreciable extent due to their high affinity towards silty soils observed at this site. (See hydrogeology discussion, Section 24.2.3). The measured soil hydraulic conductivity at this site ranges from 1.5×10^{-6} to 2.64×10^{-4} ft/sec, (Table 4-4) further indicating the retardation of movement of PCBs and lead.

Based on the detection of only minor contaminant residues in the ground water, and, more importantly, on what is known regarding their behavior in ground water, the ground water transport route is not considered a significant means of transporting PCBs and lead to offsite locations. Because residues of the other monitored components such as cadmium were not detected in the ground water, and there are no ground water users in this area, the ground water transport route will not be considered further as a component of the risk assessment.

24.4.1.3 Receptor Evaluation

Human

As described in the section on the general land use within and adjacent to the Refuge, the area within the Refuge is not populated by humans. There are only minor numbers of humans occupying the Refuge for industrial (occupational) purposes, and the Refuge has only a moderate daytime recreational use load. A large proportion of the refuge is designated as a wildlife sanctuary, and is therefore posted off limits to human use.

The area in the direct vicinity of the Job Corps site is not used as a manufacturing area. There are no users of ground water in this area. Access to the landfill has been minimized by cultivating tall and thick brush and trees in the area. Therefore, exposures by humans on a day to day basis would not be expected.

However, because of the existence of open water at this location, and its attractiveness to recreational users of the Refuge, a small number of human receptors could potentially experience potential one-time or isolated multiple exposure. This area of the Refuge is open for deer hunting for only one week per year; it is closed to the public for the remainder of the year. On this basis a human receptor could be characterized as a hiker or hunter. Also included as a human receptor would be a Refuge worker who might visit the area as part of a routine maintenance of the spillway.

It is likely that the total number of human receptors is low. A upper bound estimate of the number of human use-days in the pond area might be on the order of 25 per year according to the Refuge Manager. One human use-day is defined as a period of about four hours, where the pond and adjacent areas are used by

one human for recreational purposes. However, because the adjacent areas are more attractive to recreational users, the actual landfill area would experience a much lower amount of use days. An upperbound estimate of human use in the landfill, as related to hiking through the landfill on the way to a more "attractive" recreational area might be 10 use-days per year. Specific scenarios under which the human users of the area might become exposed to the PCBs, cadmium or lead residues in the area will be developed in the following sections.

The transport route evaluation identified three major functional transport mechanisms: the air route, the direct contact route, and the surface water transport route. Human recreational receptors identified in the previous section would be within the influence of each of these transport routes. However, for the most part, they would only be able to experience exposures while within the area of the landfill and adjacent pond, or as a consequence of their use of that area. There is no route for ingestion of contaminated fish unless a human receptor was to illegally catch and consume fish from the pond. Humans at areas removed from the landfill and pond, such as residents of nearby communities or other populated areas, as well as recreational users of the Refuge not entering the landfill or pond, would not be within the influence of the site.

The following sections describe the most likely exposure scenarios associated with each of the identified functional transport routes.

a) Direct Contact: The most likely human exposure scenario would be exposures to PCBs, lead, cadmium, and other waste components as a

result of direct contact with the waste materials and contaminated pond sediments. Because there are no human populations residing in the areas of the landfill, chronic (repeated long term) exposures would likely not take place. Direct contact exposures would occur on a one-time, or limited multiple event short term basis, experienced by recreational users of the pond and adjacent landfill.

Although PCBs and lead are not especially permeable through intact human skin, the most likely route by which they could enter the receptor's body following exposure would be through incidental ingestion of the soils or vegetation on the receptors body. Direct contact exposures could also extend into the users' homes, providing exposures to secondary receptors such as family and friends. It is possible that waste residues might become adhered to the shoes or clothing of a recreational user, to be transported to the user's home and become established as residues. The introduction of contaminated materials into a home along with soiled clothing or shoes could establish a reservoir of material that would persist and represent a longer term, chronic exposure.

b) Air Route: Dusts generated by wind erosion or foot traffic, as well as volatilized residues, represent a source and pathway for exposures via the air route. The most likely human receptors experiencing exposures via this route would include hikers or recreational users who would disturb and breathe contaminated dusts kicked up from the ground or dislodged from broad-leaf vegetation while walking through the landfill or adjacent areas. These exposures are likely to occur on an acute (one-time) basis or on a few occasions. Due to the absence of human populations near the

site, no chronic exposures would be expected. However, long distance transport of dusts towards populated areas is unlikely due to the level of vegetation at the site. Also possible, would be the inhalation of dusts by the recreational users and secondary receptors such as family members, who breathe dusts arising from soils adhered to the boots and clothing of the recreational user and transported out of the landfill.

c) Surface Water Route: The major uptake route usually associated with surface water exposures, such as by swimming and wading, is the inadvertent ingestion of water and sediment. However, the surface water route would not be a major human exposure route at this location because the pond is not used by humans for swimming or wading. There are other areas of the Refuge, notably Crab Orchard Lake, which are more attractive for swimming and wading. Nevertheless, it is possible that contact with the pond water and sediments might occur on a less than whole body basis, as might be associated with fishing and hiking. Neither fishing nor hiking are practiced in the immediate area. This route will be further evaluated within the quantitative assessment.

d) Ingestion: Ingestion of fish and game in the Job Corps landfill or the adjacent pond by human recreational users is an exposure pathway which requires further consideration. PCBs are lipophilic compounds which tend to partition into and accumulate in fat-containing tissues of animals. Ingestion of fish and game by humans utilizing the area near the Job Corps landfill would therefore be a potential scenario of exposure and uptake of PCBs as well as lead. However, only lead, and traces of cadmium and mercury were

found in a limited sampling of pond fish; PCBs were not detected in this sampling. A 1982 survey of contaminants in deer tissue at the Refuge (Ruelle, March 1983) did not show measurable PCB levels in either the fat or red meat tissues analyzed. A 1980 survey (Gritman, 1982) detected average lead levels in deer liver tissue at 5.6 mg/kg.

A relatively large percentage of the human users of the Refuge would be within a mile or so of the area for purposes of hunting. Fishing in the Job Corps site pond is not permitted although illegal fishing is conceivable. The abundance of game fish in Crab Orchard Lake and other lakes nearby further discourages fishing at the Job Corps pond. Because game would be taken from the area only on isolated occasions, this route would not represent a chronic exposure, but would be limited to a single acute or multiple acute exposures. Due to very the low likelihood of repeated use of the pond for fishing, human exposures by the ingestion of fish will not be considered further.

Wildlife

As described in the introduction to this report, Crab Orchard National Wildlife Refuge is an area that has been set aside to foster the breeding and preservation of wildlife endemic to that part of the country. Based on field inspections of the Refuge, as well as accounts relayed by Refuge Managers, there is an abundance of wildlife in the area. Of special note are the large populations of migratory aquatic waterfowl, including various species of ducks and geese who use the Refuge as a stopover during their excursions

north. Also of note are populations of white-tailed deer and numerous species of small mammals such as raccoon, rabbit and opossum. Crab Orchard Lake is also abundant with various species of warm-water fish, such as bass and catfish.

The area in the direct vicinity of the Job Corps Landfill is similar to the other areas of the Refuge. Of special note is the pond which is directly adjacent to the landfill. Field inspection of this location noted that the pond contained species of panfish, turtles, frogs, aquatic insects, and abundant aquatic vegetation. The pond was observed to be frequented by transient waterfowl. During the inspection, roughly 30 geese carcasses were found in the area. There were also deer tracks and a number of small mammal burrows.

With the exception of the general field inspections, there has been no formal survey of wildlife conducted on the area of the landfill and the adjacent pond. According to Ruelle (1987), the Refuge contains two active nesting areas of the endangered bald eagle. It could not be determined whether, in addition to common species of endemic wildlife, there were any other threatened or endangered wildlife species in the area. However, because many of the areas of the Refuge have been set aside as wildlife breeding grounds and also as stopovers for migratory birds, it is possible that other threatened or endangered species may be present in the Refuge.

Based on the field observations, as well as the high wildlife density in other portions of the Refuge, it is concluded that the Job Corps landfill and the adjacent pond contain wildlife receptors which

warrant consideration within this risk assessment. This area contains open water attractive to water-fowl, and the pond is of adequate area and depth to support aquatic life. The landfill area has become overgrown with vines which may provide a desirable location for the establishment of dens by small mammals. Therefore, it can be concluded that the landill and adjacent pond can, and most likely do, provide a transient as well as permanent habitat for various species of wildlife.

The following sections present exposure scenarios by each of the transport routes identified in the preceding sections as functional.

a) Air Route: Wildlife in the area would experience exposures to waste components by the air route. However, in contrast to the activities described for the human exposure, wildlife would be expected to come into more intimate contact with waste contaminated dusts and vapors through burrowing, preening and feeding activities. Therefore, inhalation exposures would be likely to be of considerable duration and frequency. In contrast to the limited frequency of exposures experienced by the human receptors, wildlife would experience both acute (in the case of transient species) as well as lifetime chronic exposures (in the case of endemic species). The magnitude and significance of these exposures will be discussed in the quantitative risk assessment.

b) Direct Contact: As with the inhalation route, wildlife in the area would experience chronic as well as acute exposures to the PCBs, cadmium, and lead in the landfill and the adjacent pond. These exposures would result in prolonged skin contact with the soil-borne

wastes, as well as ingestion of the materials through preening activities.

c) Ingestion: In contrast to the acute exposures potentially experienced by human users of the area, wildlife inhabiting the site would be likely to experience both acute as well as chronic exposures to PCBs, cadmium, and lead and other waste components via ingestion. These exposures would result from incidental ingestion of soils, vegetation, or sediments during preening and dusting (birds and small rodents) activities, as discussed in the direct contact section. Ground-feeding birds and gallinaceous birds (e.g. quail) ingesting grit could also ingest quantities of contaminated soil. Exposure would also occur as a result of normal dietary ingestion of vegetation and wildlife in the area.

Soils and dusts containing waste components would be consumed by herbivores along with contaminants accumulated in terrestrial or aquatic vegetation. Many aquatic waterfowl also consume large quantities of sediment as part of their diet. These inputs could represent a major exposure and uptake route.

With regards to animal life, aquatic and terrestrial inhabitants would be expected to contain bioaccumulated residues of PCBs, cadmium and lead. These bioaccumulated residues would be consumed by predatory wildlife, resulting in ingestion exposures. A limited survey of Job Corps Pond fish residues did not, however, detect PCB residues.

24.4.2 Quantitative Assessment

24.4.2.1 Estimates of Release and Exposure Rates

Estimates of Airborne Exposures

The qualitative section of this assessment has established that the air pathway represents a complete exposure route. This pathway consists of the breathing of PCB-contaminated dusts at the Job Corps landfill site created by occasional human recreational activities (i.e., hiking, hunting) and by the burrowing activities of wildlife, as well as PCB vapors evaporating from the soil. No air monitoring surveys were conducted, (except for the purpose of worker safety) nor were appropriate models available to estimate the quantity of respirable dust kicked up or dislodged into the breathing zone of a walking human.

Hwang et. al. (1986) developed one methodological approach for modeling exposure to airborne PCB residues which have evaporated from the adsorbed or liquid state on or in soils. Using the vapor pressure as a measure of the air concentration of Aroclor 1254 and the estimated air-soil partitioning behavior of PCBs, these authors reasoned that, for example, soil concentrations of Aroclor 1254 in excess of approximately 4 mg/kg would be a saturation concentration. Further, under such conditions the volatilization of PCBs can be estimated purely by consideration of the vapor pressure, without consideration of the soil adsorptive properties of the residues. In addition, the mass flux would thus be independent of the PCB concentration in the soil.

Hwang et al. (1986) estimate that the emission rate for Aroclor 1254 under saturated soil conditions is 1.13×10^{-10} g/sec/cm^2, or 1.13 ug/sec/m^2. Since the soil PCB concentration at this site exceeds the 4 ug/g concentration stated to represent a saturated condition in soil, the

emission rate given above will be used to model inhalation of PCB vapors on the site. U.S. EPA (1986) presents a model for estimation of airborne residues emitted from a hazardous waste site which considers both advection and dispersion. Conceptually, the model is comparable to the methodology employed by Hwang et al. to perform a similar estimate. However, because U.S. EPA (1986) presents the methodology in greater detail, it will be utilized here. The basic relationship is:

$$C(x) = \frac{Q}{pi*y*z*u}$$

where:

C(x) = is the concentration of substance at distance x from site (mass/volume)

Q = release rate of substance from site (mass/time)

y = dispersion coefficient in the lateral (crosswind) direction (distance)

z = dispersion in the vertical direction (distance)

u = mean wind speed (distance/time)

pi = 3.141593

For the purpose of a worst case assessment, it can be assumed that an individual is standing on a down wind edge of the Job Corps Landfill, and is exposed to PCB vapors emitted from an area of 45 m x 1 m (the landfill is approximately 0.5 acre in area, or about 45m x 45m). Thus, the PCB emission rate is 45 m^2 x 1.13 ug/sec/m^2 or 50 ug/sec. Using this value to solve for C(x).

Case	u (m/sec)	Y	Z	C(x) ug/m3
1 1 hr	1	1	2.5	6.37
2 24 hr	2	2	3.5	1.14
3 7 day	3	6	5	0.18

The relative contribution of this component of airborne exposure is discussed below.

For the purposes of this analysis, it is assumed that an adult male (70 kg) is exposed to an average of 10 mg/m3 of respirable dusts and soil particles (a level considered to constitute a nuisance) kicked up during a 4-hour recreational activity at the site. The soil monitoring data at this site gave a geometric mean value of approximately 7,950 mg/kg dry weight of total PCBs in soil, with a maximum of 50,000 mg/kg wet weight (69,042 mg/kg dry weight). Also assuming an inhalation rate of 1.3 m3/hour for light activity (USEPA ECAO-CIN-477, 1985) and that all the inhaled PCB is bioavailable, a total exposure of 413 ug PCB is obtained in the 4 hour period using the mean PCB concentration, or 5.9 ug/kg body weight. In comparison, a four hour exposure to vapors of PCBs under worst case conditions as calculated using the air model above would produce a total exposure of 1.3 m3/hr x 6.37 ug/m3, or 33 ug, less than 8 percent of the component of exposure from dust inhalation. For this reason, exposure to PCBs in the vapor state was not considered further in this assessment. Assuming three 4-hour visits per year, a daily inhalation intake of 0.05 ug/kg/day is estimated.

Comparable exposure by inhalation to soil residues of N-nitrosodimethylamine at 440 ug/kg results in a daily intake of 2.6×10^{-6} ug/kg/day. The value for lead assuming the same exposure scenario and a soil level of 5,000 mg/kg would be 0.03 ug/kg/day.

The acute toxicity of PCBs and lead is quite low to humans and terrestrial animals. For instance, the acute oral LD50 of Aroclor 1254 in adult Sherman rats is 4 to 10 g/kg body weight (USEPA AWQC, 1980). Acute inhalation toxicity data were not located, but an intravenous LD50 value in the rat of 0.358 g/kg for Aroclor 1254 may be a worst case ap-

proximation. Thus, a safety margin of approximately 100 or more exists for acute inhalation exposure to PCBs of a human under reasonable worst case conditions. Lead will be considered in Section 24.4.2.2.

Because the Job Corps landfill is very rarely used for recreational purposes, chronic exposures at the site do not exist for humans. The contribution of inhaled soil particles containing PCBs and lead to lifetime health risks in humans is discussed in Section 24.4.2.2.

Unlike humans, chronic exposures of wildlife to PCBs and lead at the site are likely by inhalation and other routes. The site provides good habitat for small mammals such as mice, chipmunks, and the like which may inhale soil dusts and vapors during burrowing activities. As with humans, no models were located for estimating inhalation exposures in small mammals under these field conditions. Using an active breathing rate value of 0.0042 m3/hour for a 30 g mouse (USEPA ECAO-CIN-477, 1985) and creation of a 10 mg/m3 dust of 7,950 mg/kg dry weight PCBs during 1 hour of daily burrowing, a daily chronic inhalation exposure of 0.016 mg/kg body weight PCB is obtained. The comparable value for lead at a soil level of 5,000 mg/kg is 0.004 mg/kg. In addition, all rodents living in burrows within the contaminated area of the landfill (within the top 1 foot of surface soil) would be expected to inhale a significant portion of PCB vapors enumerating from the soil. Cadmium or lead vapors are not expected to be significant due to the affinity of these metals for soils and the low vapor pressures of these metals compounds. Assuming that the air is saturated with PCB vapors at the present conditions at the landfill, a still wind speed of 1 mph, and an estimated residence time of 16 hours in the burrow (one hour of which is spent burrowing, the remainder resting), an exposure of 1.03 mg/kg day is

estimated for this route. Table 24.1 presents additional detail for the wildlife exposures estimated for this site.

Other types of wildlife which might receive exposures via the inhalation route at this site include deer, rabbit, mink and otter; the corresponding exposure levels for each species depends on their breathing rates, body weight and habitat. The magnitude, contribution and significance of these airborne dusts to total lifetime exposure are presented in Section 24.4.2.2, Quantitative Risk Assessment Table 24-1.

Estimates of Exposures by Direct Contact

The direct contact route of exposure at the Job Corps landfill has been identified as complete for both humans and wildlife. However, since PCBs and lead are very tightly bound to soils and sediments, dermal absorption of PCBs and lead is not expected to be significant for humans. The contributions from this exposure pathway for wildlife are addressed below in the evaluation of ingestion exposures, as a result of prolonged intimate contact with contaminants in soil during daily burrowing activities. Rather, the pathway consists of ingestion of soil-bound residues picked up through direct contact with the soils. Therefore, the contribution of this route of exposure will be discussed in the following section on ingestion exposures.

Estimates of Ground Water Exposures

As previously discussed in Section 24.4.1.2, groundwater residues of PCBs and lead are minimal in the silty soils encountered at this site, and most likely resulted from the fill material during the drilling of monitoring wells. Furthermore, there are no ground water users at this site or in

the eastern portion of the Refuge. Therefore, the ground water exposure pathway is incomplete, and will not be considered quantitatively.

Estimates of Exposure by Surface Water

The qualitative assessment has identified the surface water route as complete at the Job Corps site in view of the direct contact of the landfill with the pond, the presence of PCBs and other waste components in the pond water and sediments, and the availability of an outlet stream for offsite transport of residues. Although the pond may serve as a major source of drinking water for large mammals (i.e. deer, raccoons, beaver) and waterfowl, it will not produce significant human exposures.

Surface water ingestion by wildlife will be considered in the following section on ingestion. The presence of PCB residues in the pond sediments and water column provide a source of chronic exposure by aquatic organisms and water fowl. Mean values of total PCB residues were calculated from site monitoring data to be 0.045 ug/L for pond water and 109,635 ug/kg for sediment. USEPA AWQC (1980) has established a criterion of 0.014 ug/L of PCB to be protective of freshwater aquatic life under chronic exposure conditions and 2.0 ug/L under acute exposure conditions. Thus, pond water concentrations of PCBs exceed the AWQC chronic toxicity criteria for freshwater aquatic life, but not the acute exposure level.

While there are probably too few data points to currently establish an accurate mean concentration of PCBs in the pond water, resident organisms will also receive exposure via contact with contaminated bottom sediments and ingestion of residues accumulated in food chains from the sediments. It is feasible that populations in the pond may be at risk

from exposure to PCB residues in bottom sediments, although a limited survey of pond fish did not show detectable PCB levels, and a qualitative survey of pond life (Section 24.4.1.2) would suggest that such impact is not grossly obvious.

Estimates of Ingestion Exposures

Human

PCBs as a class are highly lipophilic materials which are readily taken into the body and tend to resist metabolic destruction and elimination. Thus, the well documented ability of PCBs to bioconcentrate in organisms from ambient media and food is one of the most significant chemical and toxicological properties of these materials. The presence of PCB residues in surface soils and in the pond water and sediments presents a source of exposure via ingestion of accumulated residues of food items by terrestrial and aquatic wildlife.

As discussed in the Receptor Evaluation, human consumption of fish caught in the Job Corps Pond will be negligible, since recreational fishing at this site is not permitted, and nearby Crab Orchard Lake offers an abundance of popular game fish for the sports fishermen. In addition, a limited survey of pond fish did not show detectable level of PCBs (0.4 mg/kg detection limit). This pathway is therefore technically incomplete for humans and ingestion of fish will not be considered in the quantitative assessment of human exposures.

Human ingestion exposures to PCBs may also occur by consumption of contaminated venison taken during the limited Refuge deer season. However, as cited in Section 2.6.4 of this report, analysis of ten Refuge deer for PCB contamination showed no detectable residues in either fat or

red meat deer tissue. Given these data and the transient nature of grazing by deer at the site, the probability of human exposure appears nil.

For purposes of the evaluation of ingestion exposure, an individual is assumed as a worst case estimate to consume 100 mg of soil contaminated with 7,950 mg/kg of PCBs per trip to the site (i.e. for hunting), as the result of direct contact with the soil to skin, boots and clothing. This produces a total ingestion exposure of 0.011 mg/kg/trip for a 70 kg adult. For the worst case assumption of three such trips per year, recurring over a 70-year lifetime, a daily intake of 0.093 ug/kg/day is estimated for the ingestion route (see Table 24-1). A comparable value for lead exposure would be 0.058 ug/kg/day for ingestion of soil containing 5,000 mg/kg lead. The corresponding daily intake from ingestion of soil containing N-nitrosodimethylamine at 440 ug/kg would be 5.2×10^{-6} ug/kg/day. The potential human health risks associated with these chronic exposures are discussed in Section 24.4.2.2.

Wildlife

Data are limited for estimating the chronic effects of PCB and lead to wildlife. USEPA AWQC (1980) discussed studies indicating mortality and reproductive failure of mink fed PCB-adulterated fish meal. A nine month feeding of 2 mg/kg dietary Aroclor 1254 significantly inhibited reproduction and 10 mg/kg resulted in complete mortality of pregnant female mink. For purposes of the wildlife assessment, it is assumed that PCB concentrations in fish are present at one-half the detection limit used in the survey of pond fish (since the analyses did not measure detectable residues), for a concentration of 0.2 mg/kg. An assessment of the exposures to wildlife from pond fish is discussed in Section 24.2.2.2.

Wildlife may also receive exposure from inadvertent ingestion of contaminated soil, vegetation, or sediment via preening or burrowing activities and consumption of food at the site. The magnitude of such exposures is also a function of food preference, home range, and migratory pattern of the species. For instance, deer move relatively long distances (large home range), feed on or browse through vegetation located higher off the ground and do little preening. Rabbits have a small home range, feed on broad leaf vegetation located close to the ground that usually has higher contaminant concentrations due to rain splash, and they preen regularly. Some duck (diving) species dabble in sediments but only spend part of the year in an area and when settled they move frequently from one body of water to another. Such species would not be expected to receive significant exposures via residues in the pond sediments since these might spend approximately two months per year at the Refuge, and during this time would reside preferentially in nearby Crab Orchard Lake. On the other hand, duck species which might remain on-site long enough to receive any quantifiable exposures are predominantly vegetarian types such as mallards or other surface dwellers, and could be exposed to contaminants as a result of feeding on terrestrial and aquatic vegetation. Mice live close to the soil, preen regularly, and may consume roots or other plant parts growing close to the soil that can accumulate relatively high levels of contaminants from rain drop splash and wind erosion. In order to assess the potential risks to wildlife posed by site soil and vegetation residues, these species were chosen to represent a variety of such biological and ecological variables. The magnitude of these ingestion intakes of soil, sediment and vegetation

and their significance in a lifetime of exposure to site PCB residues is detailed in the following section on Quantitative Risk Assessment.

Exposure of small burrowing animals such as mice to soil residues is also possible by dermal absorption, as well as by direct ingestion. No guidance was located for conducting an exposure assessment of such an event. Key elements of such an assessment, such as the available body surface area of a generally fur-covered animal through which such absorption might occur and the rate of dermal transfer of soil bound PCBs are a matter of conjecture. Realizing these uncertainties, the following scenario was constructed.

It is assumed that a small burrowing rodent such as a 30 g mouse lives in intimate contact with site soils containing a level of 7,950 mg/kg PCBs. It is further assumed that the body surface area available for absorption includes close to the entire body (e.g. for species that dust bathe), and that this area is about 36 cm^2. Due to the animal's grooming habits, it is assumed that soil contacting the fur of the animal is inadvertently consumed, and that these residues are considered under the direct contact/ingestion scenario discussed above. Hawley (1985) assumed that soil dust would adhere to human skin at a rate of 3.5 mg/cm^2. Assuming a similar rate for the mouse, the exposed skin of the animal will be in continuous contact with 126 mg of soil containing 7,950 mg/kg. As reviewed in USEPA 600/6-86/002, Development of Advisroy Levels for PCB cleanup (1987), an absorption fraction of 0.05 might be used for Aroclor 1254. Assuming that the soil residues of PCBs are absorbed at this rate every 24 hour period of exposure, it can be estimated that the mouse receives an exposure to soil PCBs at a rate of 1.67 mg/kg/day.

Wildlife may also be exposed to low level PCBs via drinking the pond water. The quantitative assessment for ingestion of pond water is detailed in Section 24.4.2.2, based on a mean PCB concentration of 0.045 ug/L in pond waters, and assuming a water consumption rate of 10% of body weight per day for herbivores, and a rate of 30% of body weight for carnivores (Chew, 1965).

The pond outlet stream provides a mechanism for offsite transport of PCBs via dissolved and sediment-bound residues. However, insufficient data on downstream residue levels and potential receptors prevents exposure and risk estimates. Dilution effects should lessen the downstream exposures relative to the worst case pond situation.

Burrowing animals such as mice are estimated to ingest 1.5 grams as food (10% soil, 90% vegetation), based on a consumption of 5% of body weight. Soils at this site contain an average of 5,000 mg/kg lead, while lead levels in vegetation are assumed as 1 percent of the soil concentration, based on the fraction expected to leach from soils for uptake by site plants. An exposure of 47.5 mg/kg/day of lead is estimated as a result of daily burrowing and inadvertent ingestion of dust while grooming. Heavy metals may also bioaccumulate in organisms such as earthworms which ingest soil while feeding. In particular, cadmium has been demonstrated to accumulate in earthworms by a factor of 50-fold over soil concentrations (Ruelle, 1987). This creates the potential for ingestion exposures in wildlife via contamination of food chains, as well as the pond. The significance of these exposures is discussed in the following section.

24.4.2.2 Quantitative Risk Assessment

The estimates of total PCB intakes of receptors in complete exposure pathways are summarized below:

Human Risks

Human PCB exposure at the Job Corps site is technically acute in nature rather than chronic and is much lower than any which would raise concerns for acute PCB toxicity. For instance, access to the Refuge for deer hunting is granted on a limited basis, greatly reducing the probability for repeated, chronic exposures as assumed in the following worst case estimate. The chronic PCB toxic effect of most concern is carcinogenicity as evidenced by lifetime dietary studies in rodents. The risk of excess cancer caused by human direct contact exposures at the Job Corps site may be estimated by multiplying the average daily intake of PCBs estimated in Table 24-1 by the PCB unit cancer risk factor (i.e. potency factor; a dose-response factor derived from a conservative mathematical extrapolation of the animal cancer data). For PCBs, the unit cancer risk factor determined by EPA is 7.7 $(mg/kg/day)^{-1}$ (Exhibit A). Therefore, the cancer risk estimate for humans due to PCB exposure is $7.7 \times 1.46 \times 10^{-4} = 1.1 \times 10^{-3}$.

An additional potentially carcinogenic substance, N-nitrosodimethylamine, was also present at this site. However, the quantitative data for this compound is based only on a single detection in soil during the Phase I investigation. Current risk assessment methodology suggests that risks from concurrent exposures to a mixture of carcinogenic substances can be approximated by addition of the risk estimates of the individual substances. The exposure assessment determined that, under the assumed exposure conditions, using twice the level detected at the site, N-nitrosodimethylamine residues in the soil might result in a daily intake

TABLE 24-1 (p. 1)

SECTION 24.4.2: JOB CORPS LANDFILL & POND
QUANTITATIVE RISK ASSESSMENT

SPECIES	BODY WEIGHT kg	ESTIMATED DAILY PCB INTAKE												
		Inhalation Rate (1)				Food (2)			-Pond Water (3)-		-Pond Fish (4)-		-Dermal (5)-	TOTAL
		air m3/hr	dust ug PCB	vapor ug PCB	PCB ug/kg	Soil grams	Veget/ Insects grams	PCB ug/kg	Liters	PCB ug/kg	Fish grams	PCB ug/kg	Absorption mg/kg PCB	EXPOSURE mg/kg/day
MAN	70	1.3	3.40	0.27	0.05	0.10	~0	0.093	~0	~0	~0	~0	~0	1.46E-04
Deer	60	1.3	413	33	7.44	170	1530	24,552	6	0.270	-	-	~0	24.56
Mallard	1	-	~0	~0	~0	5	45	42,523	0.1	0.0045	~0	~0	~0	42.52
Rabbit	1	0.083	6.63	~0	6.63	7.5	67.5	64,991	0.1	0.0045	-	-	0.01	65.01
Mouse	0.03	0.0066	0.48	33	1,104	0.15	1.35	43,328	0.003	0.000135	-	-	1.67	46.10
Mink	1	0.083	6.63	~0	6.63	~0	75	5,963	0.3	0.0135	75	15	~0	5.98
Heron	3	0.25	20	~0	6.63	~0	~0	~0	0.9	0.0405	350	23	~0	0.030
Otter	9	0.75	60	~0	6.63	~0	~0	~0	2.7	0.1215	900	20	~0	0.027

~0 = negligible contribution
(-) Not applicable or irrelevant
(#) Active breathing rate used for dust exposure. Resting rate of 0.0015 m3/hr used for vapor exposure.
Additional assumptions listed on Page 2.

TABLE 24-1 (p. 2)

ASSUMPTIONS

(1) INHALATION EXPOSURE

Dust Inhalation: Based on 10 mg dust inhaled per m3 of air, 7.95 ug mean PCBs per mg dust, and 1 hr (small mammals) or 4 hrs (man and deer) exposure duration each contact.

Vapor Inhalation: Exposures for man and deer estimated based on air model by Huang, et.al. (1986); see text. Vapor exposures for burrowing animals based on saturation concentration of Aroclor 1254 (vapor pressure = 1.49E-06 psia) resulting in approx. 1.36 mg PCB/m3 air, species breathing rate, and estimated duration in burrow (16 hrs for mouse).

(2) INGESTION EXPOSURE

Total Food (Vegetation + Soil + Terrestrial mammals/insects):
Intake for wildlife calculated as 5% of body weight, except deer (1.7 kg/day, Mulsey and Ulrey, 1982), rabbit (75 g/day, Green and Dunsmore, 1978), and mink (150 g/day, 50% as fish (Towell and Tabor, 1982; G. Smith, 1988), and 50% as terrestrial insects/mammals (Gerould, 1988). Diet for mallard reflects preference of 75% terrestrial insects and 25% vegetation in consideration of one of the most susceptible periods of their life cycle. Mink and mallard total food exposures are adjusted for home range; the affected area is estimated to comprise 10% of the home range for these species.

Soil/Sediment: Ingestion rate based on mean PCB concentration of 7.95 ug/mg soil and inadvertent consumption of particulates while preening or burrowing (wildlife), or ingested as a result of soiled clothing or boots (man).

Vegetation Concentration of PCBs estimated as 1% of mean soil concentration or 7.95 mg/kg.

Terrestrial Animals PCB concentration assumed to be 10% of residual soil concentration (i.e. 795 mg/kg). (Gerould, 1988).

(3) Water intake rates assumed to be 10% of body weight for hervibores, 30% of body weight for carnivores, and a mean pond water concentration of 0.045 ug/L.

(4) Fish and aquatic foods are assumed to contain PCB residues at one half the analytical detection limit based on a limited survey of fish which did not show detectable PCB residues (0.4 mg/kg detection limit). Mink may consume 50% or 75 g/day of their total diet as fish (Towell and Tabor, 1982; G. Smith, 1988). Otter are estimated to consume 900 g fish/day (90% of 1 kg food/day, Chapman and Feldhamer, 1982); great blue herons consume 350 g food/day (Kendeigh, 1970; Kushnan, 1977) consisting mostly of fish and aquatic organisms.

(5) DERMAL ABSORPTION

Dermal absorption exposure would be negligible for transient, large terrestrial mammals such as humans or deer. Absorption exposure would be negligible for species which bathe frequently in water, or otherwise spend little time in direct contact with soil (e.g. mink, heron, otter, mallard). Exposure estimates for rabbit and mouse are based on approx. body surface area available for constant contact (10 cm2 for rabbit and 36 cm2 for mouse), adhesion of 3.5 mg soil per cm2 of body (Hawley, 1985), and 5% extent of absorption for Aroclor 1254 (EPA/ORD, 1986).

of 5.2×10^{-6} ug/kg/day. USEPA (1987) estimated that a daily intake of 0.0137 ug of this compound may be associated with a risk of excess cancer of 10^{-5}. Thus, assuming that the 440 ug/kg soil residue of nitrosamines is representative of the entire site, a cancer risk of 2×10^{-7} may be posed by site exposures to N-nitrosodimethylamine. Adding this risk to the risk estimated for PCB exposure at the site, a total carcinogenic risk of 1.1×10^{-3} is estimated.

This level of risk is higher than the 10^{-7} to 10^{-4} population risk level generally considered by regulatory and public health agencies to pose minimal excess health concerns on a national basis. This estimate used very conservative values which are mitigated by several major factors discussed in Section 24.4.3, Analysis of Uncertainties.

Total human intake of lead was estimated to be 0.088 ug/kg/day for this scenario. For comparative purposes, the USEPA OHEA (1987) has determined an acceptable daily intake of lead to be 1.43 ug/kg/day. Thus, the estimated upper bound lead exposure possible at this site is well below the acceptable daily intake level of 1.43 ug/kg/day.

Cadmium is a human carcinogen by the inhalation route and produces serious harm to the kidney at low levels of chronic ingestion. Its carcinogenic potency is estimated to be greater than that of PCBs (Exhibit A). Therefore, chronic exposure to soil residues of cadmium would be of great concern if it were to be shown that it was present at consistently high levels at this site.

Wildlife Risks

The mean water concentration of PCB in the pond is below the criterion of 2 ug/L considered by USEPA AWQC (1980) to protect

TABLE 24-2 (p. 1)

JOB CORPS LANDFILL & POND
EXPOSURE ASSESSMENT FOR 50 MG/KG CLEANUP

SPECIES	BODY WEIGHT	ESTIMATED DAILY PCB INTAKE												
		Inhalation Rate (1)				Food (2)			Pond Water (3)		Pond Fish (4)		Dermal (5)	TOTAL
	kg	air m3/hr	dust ug PCB	vapor ug PCB	PCB ug/kg	Soil grams	Veget/ Insects grams	PCB ug/kg	Liters	PCB ug/kg	Fish grams	PCB ug/kg	Absorption mg/kg PCB	EXPOSURE mg/kg/day
MAN	70	1.3	0.02	0.27	0.0042	0.10	~0	0.001	~0	-	~0	~0	~0	4.77E-06
Deer	60	1.3	3	33	0.593	170	1530	154	6	-	-	-	~0	0.155
Mallard	1	-	~0	~0	~0	5	45	267	0.1	-	~0	~0	~0	0.267
Rabbit	1	0.083	0.04	~0	0.04	7.5	67.5	409	0.1	-	-	-	0.0001	0.409
Mouse	0.03	0.006#	0.003	7.20	240	0.15	1.35	273	0.003	-	-	-	0.011	0.523
Mink	1	0.083	0.04	~0	0	~0	75	37.5	0.3	-	75	15	~0	0.053
Heron	3	0.25	0.13	~0	0.042	~0	~0	~0	0.9	-	350	23	~0	0.023
Otter	9	0.75	0.38	~0	0.04	~0	~0	~0	2.7	-	900	20	~0	0.020

~0 = negligible contribution
(-) Not applicable or irrelevant
(#) Active breathing rate used for dust exposure. Resting rate of 0.0015 m3/hr used for vapor exposure.
Additional assumptions (notes) listed on Page 2.

TABLE 24-2 (p. 2)

ASSUMPTIONS

(1) INHALATION EXPOSURE

Dust Inhalation: Based on 10 mg dust inhaled per m3 of air, 0.05 ug mean PCBs per mg dust, and 1 hr (small mammals) or 4 hrs (man and deer) exposure duration each contact.

Vapor Inhalation: Exposures for man and deer estimated based on air model by Huang, et.al. (1986); see text. Vapor exposures for burrowing animals based on emission rate of 0.61 ug/m3 at 1 mg Aroclor 1254/kg and 10 mph wind, corrected for 50 mg/kg mean concentration and low air turnover within burrow space (1 mph wind), for a modified emission rate of 305 ug/m3. Estimated duration in burrow 16 hrs (for burrowing rodent).

(2) INGESTION EXPOSURE

Total Food (Vegetation + Soil + Terrestrial mammals/insects):
Intakes for wildlife calculated as 5% of body weight, except deer (1.7 kg/day, Mulmey and Ulrey, 1982), rabbit (75 g/day, Green and Dunsmore, 1978), and mink (150 g/day, 50% as fish (Towell and Tabor, 1982; G. Smith, 1988), and 50% as terrestrial insects/mammals (Gerould, 1988). Diet for mallard reflects preference of 75% terrestrial insects and 25% vegetation in consideration of one of the most susceptible periods of their life cycle. Mink and mallard total food exposures are adjusted for home range; the affected area is estimated to comprise 10% of the home range for these species.

Soil/Sediment: Ingestion rate based on mean PCB concentration of 0.05 ug/mg soil and inadvertent consumption of particulates while preening or burrowing (wildlife), or ingested as a result of soiled clothing or boots (man).

Vegetation Concentration of PCBs estimated as 1% of mean soil concentration or 0.5 mg/kg.

Terrestrial Insects PCB concentration assumed to be 10% of residual soil concentration (i.e. 5 mg/kg). (Gerould, 1988)

(3) Residual PCB levels in the pond water would be non-detectable following remediation. Due to the lack of a waste source, this pathway is technically incomplete.

(4) Fish and aquatic foods are assumed to contain PCB residues at one half the analytical detection limit based on a limited survey of fish which did not show detectable PCB residues (0.4 mg/kg detection limit). Mink may consume 50% or 75 g/day of their total diet as fish (Towell and Tabor, 1982; G. Smith, 1988). Otter are estimated to consume 900 g fish/day (90% of 1 kg food/day, Chapman and Feldhammer, 1982); great blue herons consume 350 g food/day (Kendeigh, 1970; Kushnan, 1977) consisting mostly of fish and aquatic organisms.

(5) DERMAL ABSORPTION

Dermal absorption exposure would be negligible for transient, large terrestrial mammals such as humans or deer. Absorption exposure would be negligible for species which bathe frequently in water, or otherwise spend little time in direct contact with soil (e.g. mink, heron, otter, mallard). Exposure estimates for rabbit and mouse are based on approx. body surface area available for constant contact (10 cm2 for rabbit and 36 cm2 for mouse), adhesion of 3.5 mg soil per cm2 of body (Hawley, 1985), and 5% extent of absorption for Aroclor 1254 (EPA/ORD, 1986).

freshwater aquatic species from acute toxicity, although high PCB levels in the sediments may pose an undefined acute toxicity risk to benthic organisms. However, the ambient water quality criteria for chronic effects in freshwater aquatic organisms (0.014 ug/L) is exceeded by at least three-fold in the Job Corps Pond. While a qualitative examination of the pond showed no gross evidence of ecosystem impairment, it is possible that sensitive elements of the pond community are being affected by chronic PCB exposure.

Exposures to PCBs for several typical terrestrial wildlife species were evaluated for the site. For comparative purposes, a 1.0 kg adult female mink consuming 150 g daily of a 2 mg/kg PCB diet receives a daily intake of 300 ug/kg/day which is a chronic effect level for this species. The mink, however, may be unusually sensitive to PCB effects relative to other mammals.

The most obviously affected species may be fish-eating mammals (mink, otter, raccoon) and birds (herons, merganser ducks, osprey), even though a limited survey of Job Corps pond fish did not detect PCBs. If these species showed the same sensitivity to PCBs as shown by mink and chickens in laboratory studies, reproductive failure and possibly overt lethality may occur at this site. The estimated whole fish PCB concentration of 12.5 mg/kg could potentially exceed the chronic dietary levels (2-5 mg/kg) producing lethality and reproductive failure in mink, a piscivorous mammal (USEPA AWQC, 1980). Even using less conservative estimates for BCF and water PCB concentrations, sensitive piscivorous mammals such as mink may be at risk from pond PCB contaminants, provided they accumulated to the levels estimated based on their physical properties.

Fish-eating birds such as herons and certain ducks may also be at risk but insufficient data exist to estimate degree of risk. Of particular concern is the location of two active bald eagle (an endangered piscivorous species) nests on the Refuge, since fish from the site pond might be a source of prey (Ruelle, 1987). Domestic chickens given 20 mg/kg dietary PCB displayed a broad spectrum of reproductive and teratogenic effects (Exhibit A). A quantitative assessment of exposures to bald eagle due to ingestion of PCB residues in fish is presented in Section 38.4, Wildlife Assessment for Crab Orchard Lake. No PCBs were detected in fish sampled at the Job Corps Pond, and, due to the large foraging range of these species, exposures to bald eagles at this would not be expected to be significant. For other species such as deer and duck, incidental, short term exposure to pond sediments while browsing or searching for food are relatively low based on the site conditions and habitat (see Table 24-1).

USEPA ECAO-CIN-414 (1987) determined that a daily exposure of rats to 1 mg/kg/day of Aroclor 1254 constituted a subchronic no adverse effect level in that species, compared to the site ingestion exposure estimate for mice of 46 mg/kg/day. Rabbits are estimated to receive total exposures on the order of 65 mg/kg/day at this site. Thus, small herbivorous mammals such as mice and rabbits are likely to be at risk from site PCB residues.

An alternate assessment of wildlife risks from exposure to PCBs was developed independently by the U.S. Fish and Wildlife Service, which is presented in its entirety in Exhibit D. Using a three-phase fugacity model and dietary/dermal exposure assumptions, total daily PCB exposure was estimated for a burrowing animal such as a pocket gopher. As

developed, approximately 90 percent of the total exposure is determined by inhalation of PCBs in the animal's burrow. The air concentration is related to soil PCB concentration using a partitioning model based on the estimated fugacities of PCBs in air, soil, and water, as developed by Mackay and colleagues (Mackay, 1985).

It is assumed that the exposed animal spends all of its time in a sealed burrow, actively burrowing 9 hours per day and resting 15 hours per day. It is assumed that the PCB concentrations in air, soil, and water are in thermodynamic equilibrium at all times. Soil concentrations were back-calculated to represent concentrations which would result in exposures comparable to food consumption rates employed in laboratory tests for wild and domestic rodent species, which were associated with biological and toxicological effects. Two examples are presented:

1) 0.7 mg/kg soil PCBs would result in exposures comparable to the dietary PCB exposures in an experiment which was found to "... increase the liver weights in F1 male weanling rats..." and to "... decrease the circulating levels of adrenal cortex hormone B...".

2) 7 mg/kg to 14 mg/kg soil PCBs would result in exposures comparable to the dietary PCB exposures in an experiment which was found to "... decrease the weight of reproductive organs, growth rates, and reproductive success of second generation white-footed mice...".

A soil sample analyzed in the Phase I survey of the Job Corps site showed a trace level (0.22 mg/kg wet weight) of N-nitrosodimethylamine. In order to provide a conservative estimate of the potential effects from

this contaminant, and due to deficiencies noted in the analytical result for this soil sample, the quantitative evaluation for humans as well as for wildlife is based on twice the detected level or 440 ug N-nitrosodimethylamine per kg of exposed soil. Because of the intimate contact which small burrowing mammals may have with soil, the risks of direct contact of wildlife to these residue levels are assessed below, based on available information on the effects of nitrosoamines summarized in the risk assessment for Site 19 (see Section 26.4).

Using the assumptions given above for exposures of burrowing mice (30 g body weight) to site PCB and lead residues, exposure to soil residues and vegetation (assumed to contain 1% of the soil concentration) of 0.44 mg/kg N-nitrosodimethylamine will produce a daily intake of 4.2 x 10^{-3} mg/kg/day by the ingestion route. Vapor inhalation would not be expected to be significant at the low microgram level of contaminant observed. Inhalation of contaminated dusts (e.g. during burrowing) are estimated at 8.8 x 10^{-7} mg/kg/day. Using the unit risk factor of 26 $(mg/kg/day)^{-1}$ derived in USEPA ECAO (1986--) based on rat studies, and assuming a similar sensitivity for wild rodent species, a cancer risk estimate of 1 x 10^{-1} is derived for the total potential exposure received by burrowing rodents at this site. As discussed in Section 26.4, the significance of this risk level is uncertain. The effects of cancer, generally incurred later in an exposed organism's lifetime, may be very small when considered in light of other factors influencing whether a wildlife population can maintain itself (i.e. survival to reproductive age, competition, weather, disease, predation, etc). On this basis, Newell et al. (1987) chose a risk level of 10^{-2} as a level of acceptable carcinogenic risk for wildlife, with the acknowledgement that more study is needed to

justify this choice. Using this rationale, it is concluded that wildlife exposure to site residues of N-nitrosodimethylamine could result in a carcinogenic response, but the biological significance of the response cannot be assessed with currently available information.

Additional studies were reviewed to further evaluate wildlife exposures to N-nitrosodimethylamine at this site. USEPA ECAO (1986--) used a reproductive effects study in adult female mice to establish a lowest observed effect level (LOEL) of 0.019 mg/kg/day for N-nitrosodimethylamine. The estimated daily exposure rate of 0.0042 mg/kg/day at this site is below this LOEL. Thus assuming exposed species are of equivalent sensitivity to N-nitrosodimethylamine as laboratory mouse strains, the estimated exposure rates are below a level which may elicit a toxic response under subchronic conditions.

As estimated in the previous section, burrowing mammals and other site wildlife may be endangered by chronic exposure to site lead levels of 47.5 mg/kg/day. The effects of these lead residues on wildlife species are largely undocumented, but using the human criterion for acceptable intake levels, wildlife ingestion of even a small fraction of this value may produce reproductive impairment and possibly other subtle effects which might decrease survival of wild populations. A similar argument may hold for aquatic organisms and piscivorous species exposed to lead via aquatic foodchains, although insufficient data exists to assess an exposure/response relationship.

Cadmium is also potentially toxic to fish and wildlife, and can biomagnify to potentially toxic levels in soil-ingestion foodchain animals such as earthworms. Small predaceous birds and mammals would be at particular risk if cadmium was consistently elevated at this site.

24.4.3 Analysis of Uncertainties

As discussed in Section 6.5, most approaches to quantitative risk assessment are inherently conservative in order to be most protective of public and environmental health in the face of numerous scientific uncertainties and insufficiencies in case-specific data. At the Job Corps landfill site, worst case estimates indicated that risks to environmental populations may be posed by PCBs and nitrosoamines residues at the site, particularly by inhalation and ingestion of dust-borne residues. Several areas of uncertainty exist, however, which may serve to mitigate the quantitative degree of risk. First, the level of contaminants in fish consumed by wildlife were estimated using reasonable worst case assumptions on bioconcentration potential. The estimates of exposure to wildlife due to ingestion of fish were based on an average potential bioconcentration factor, although a limited survey of pond fish did not show detectable PCB levels. The fish data were used only qualitatively to support the conservative nature of the wildlife assessment, since only two samples were collected following the Phase II sampling effort.

The worst case human cancer risk assessment for PCBs and for chronic toxicity from lead was conducted using an assumption of lifetime visits (i.e. for hunting), a highly improbable event. It is not likely that the same individual will continue to hunt on this site annually, three times each year over an entire 70-year lifetime. The assessment developed for N-nitrosodimethylamine was based on doubling the result from a single soil sample analyzed in the Phase I survey, although the purpose of that survey was to screen the site and not to support a quantitative assessment. Thus, the limited analytical data for nitrosoamines do not provide for a confident estimate of the risks associated with nitrosamines

exposure. Furthermore, the quantitative assessment model that was used assumes that there is no exposure level for PCBs which does not pose a risk from cancer. However, there is accumulating evidence that PCBs may induce cancer in animal tests through a threshold mechanism, such as promotion of pre-existing lesions (Williams and Weisburger, 1986). Thus, it is possible that low exposures to PCBs may not present concerns for carcinogenic effects.

Finally, the size of the potentially exposed human population is at best very small and it is unlikely that an increased occurrence of cancer at even a 10^{-3} risk level could be detected above background levels of disease, since the probability that even one individual would meet the worst case exposure criterion is almost nonexistent. Assessment of human and wildlife risks from lead exposure is limited by lack of knowledge of the types of insoluble lead salts (i.e. sulfate, oxides) likely to be present at the site. It has been speculated that the geese found in 1985 at this site (See Section 24.1) may have died as a consequence of contaminants encountered here. However, given the limited data base, it is not appropriate to implicate site contamination as the reason for geese kills at the site, since no PCBs or other organic were detected in the carcasses and metals were not analyzed. It is of interest to note that some of the better documented cases of wildlife lead toxicity concern waterfowl (Exhibit A) presumably due to ingestion of lead shot while feeding in aquatic environments subject to hunting.

24.5 Preliminary Remedial Alternatives

On the basis of the above assessment, it was determined that the levels of contaminants present pose risk levels to exposed human and

wildlife receptors. The major contaminants of concern in soils, sediments and waters were identified to be cadmium, lead and PCBs. The explosive residue nitrobenzene was also detected in both ground water and surface water at levels below the AWQC. The contaminants in soils were found to be associated with only the top 0-1 ft. of soil in the landfill, while subsurface soils (between 1 and 3 ft. depth) did not contain detectable concentrations of contaminants with the exception of one core which showed 11.6 mg/kg PCBs and 219 mg/kg lead. Contamination in sediments was detected in the surface samples (109.6 mg/kg wet wt PCBs to 1 ft depth), but no subsurface sediments were collected to verify the levels in subsurface sediments. Fish samples from the pond showed lead levels up to 6.9 mg/kg (bluegill only), but PCBs were undetected (0.4 mg/kg detection level) and only traces of cadmium and mercury were found.

In its current condition, the Job Corps Site should not be used for any activities which would increase the potential for human and/or wildlife exposure via the water or direct contact with soils/sediments. Future uses of the site might be considered, subject to additional testing to assure the site does not constitute a risk to potential human or wildlife receptors.

Risk levels from site contaminants may be reduced to a range of 10^{-6} to 10^{-5}, generally considered acceptable by regulatory and health agencies, by implementation of appropriate remedial actions. In Section 24.4.2.2, it was estimated that PCB soil concentrations of 7950 mg/kg presented an excess cancer risk of 1.1×10^{-3} for the assumed scenarios of exposure. Using similar assumptions, the risks to humans would be reduced to an acceptable range of 10^{-6} to 10^{-5} by precluding exposure to soils containing PCB concentrations greater than 7 to 70 mg/kg. Section

24.4.2.2 also indicates that the most sensitive wildlife species, mink, shows reproductive lowest observed effects with chronic exposure to PCBs at 640 ug/kg/day (Newell, 1987). The 7 to 70 mg/kg level of remediation of exposed surface soils is 14 to 1.4 fold lower than the no observed effect level for protection of the most sensitive wildlife species. Exposure estimates under an example cleanup scenario of 50 mg/kg PCBs in soil are presented in Table 24-2. The assumptions used in calculating such exposures are similar to those used in the site risk assessment (Section 24.4).

In general, the objectives of the remedial program to be developed in the FS will be to render incomplete all possible transport routes between contaminant sources of concern and potential receptors, including the routes of direct contact (absorption, ingestion), surface water transport, and inhalation of contaminated dusts or vapors. With respect to contaminated transport pathways such as drainage ditches (as opposed to contamination sources) identified as a concern, such remedial measures as surface excavation, capping, regrading, revegetating, and surface water diversion will be emphasized in the FS.

Table 2 of the Executive Summary section summarizes the remedial responses which are likely to be the focus of the FS investigation. Some of the potentially applicable remedial measures for this site are discussed below.

Limited Site Access

One of the immediate measures to be taken at Job Corps may be to limit human and wildlife exposure to the site. Fencing and closing the area to all but Refuge Personnel, and maintaining a thick vegetative

cover, may be appropriate until further remedial action can be commenced. Deed restrictions might be imposed to limit future uses of the area.

Surface Water Control

The purpose of surface water control would be to prevent run-on and run-off within the landfill, to preserve the vegetative cover, and prevent transport of contaminated soil to the pond and eventually to Crab Orchard Lake.

Off-Site Removal or On-Site Containment of Soil and Sediment

Contaminated surface soil and sediments (up to 1-1.5 ft depth) might be excavated and removed for treatment off-site or regraded and contained on-site. Clean soil will be used for fill and capping any areas requiring excavation. Alternatives for containment of wastes might include removal to a secure landfill or secure storage such as in the Munitions Bunkers in Area 13. An estimated 700 CY of soil from the landfill area may require removal or containment based on the sampling results from the RI. The areal extent of contamination at the landfill may extend north and east beyond the area sampled an additional 25-50 ft. on either side; if this is confirmed by pre-remediation sampling, the total volume of soil for removal could be greater than 700 CY.

The sediments from the shallow areas of the pond adjacent to the landfill contained PCBs, cadmium and lead concentrations which exceeded the levels detected at the control sites. These sediments pose a concern in that contaminants can slowly leach or suspend with sediments in the water and increase the probability of exposure to wildlife and humans or

could be transported off-site. Based on the sampling program, the extent of sediment contamination in the pond is estimated to include an area extending 25 ft. from the shoreline surrounding the landfill up to a depth of 2 ft. However, this extension might be better defined prior to actual cleanup. Based on the RI data, an estimated 560-600 CY of sediment may require removal. The sediment samples collected from the deep (middle) portion of the pond did not contain elevated levels of contaminants.

The required depth and areal distribution of contaminated sediments and soil requiring cleanup might be further defined through additional sampling. The required sampling efforts might be incorporated in the field efforts proposed for the FS.

Monitoring

The remedial response alternative to be implemented at this site might include periodic sampling and analyses of the four monitoring wells and of the pond water and sediment for cadmium, lead, and PCBs. Follow-up studies might begin immediately after remediation and continue periodically to verify the adequacy of the cleanup.

24.6 Conclusions and Recommendations

It can be concluded that the Job Corps Landfill is impacted, with the primary pollutants being PCBs and lead. Based on a quantitative risk assessment, it was found that, since human exposure is limited, due to the location of the site, concerns for protection of wildlife would be the focus of the remediation effort.

SECTION 25 - SITE 18, AREA 13 LOADING PLATFORM

25.1 Site Description

Area 13 of the Refuge consists of approximately 85 bunkers that were originally built for storage of 500 lb. bombs. Most of the bunkers are currently used by Olin Corp. and U.S. Powder to store explosives. Agricultural fields are cultivated between the bunkers. This area was served by a rail spur which was abandoned and dismantled. It was reported to the Refuge Manager that chemicals used in munitions manufacturing were dumped off the platform.

Site 18 consists of the Area 13 loading platform, a concrete pad 235 feet long by 10 feet wide elevated by about five feet. (See Figure 25-1). The dock is supported on concrete posts spaced about 9 feet apart. The northwest side of the platform contains stone bedding (probably from the old railroad bed) with a number of small areas of ponded water. No unusual vegetation changes were detected. The only curious item was a pile of dirt and stone rubble off the west end of the dock with a rusted drum shell nearby.

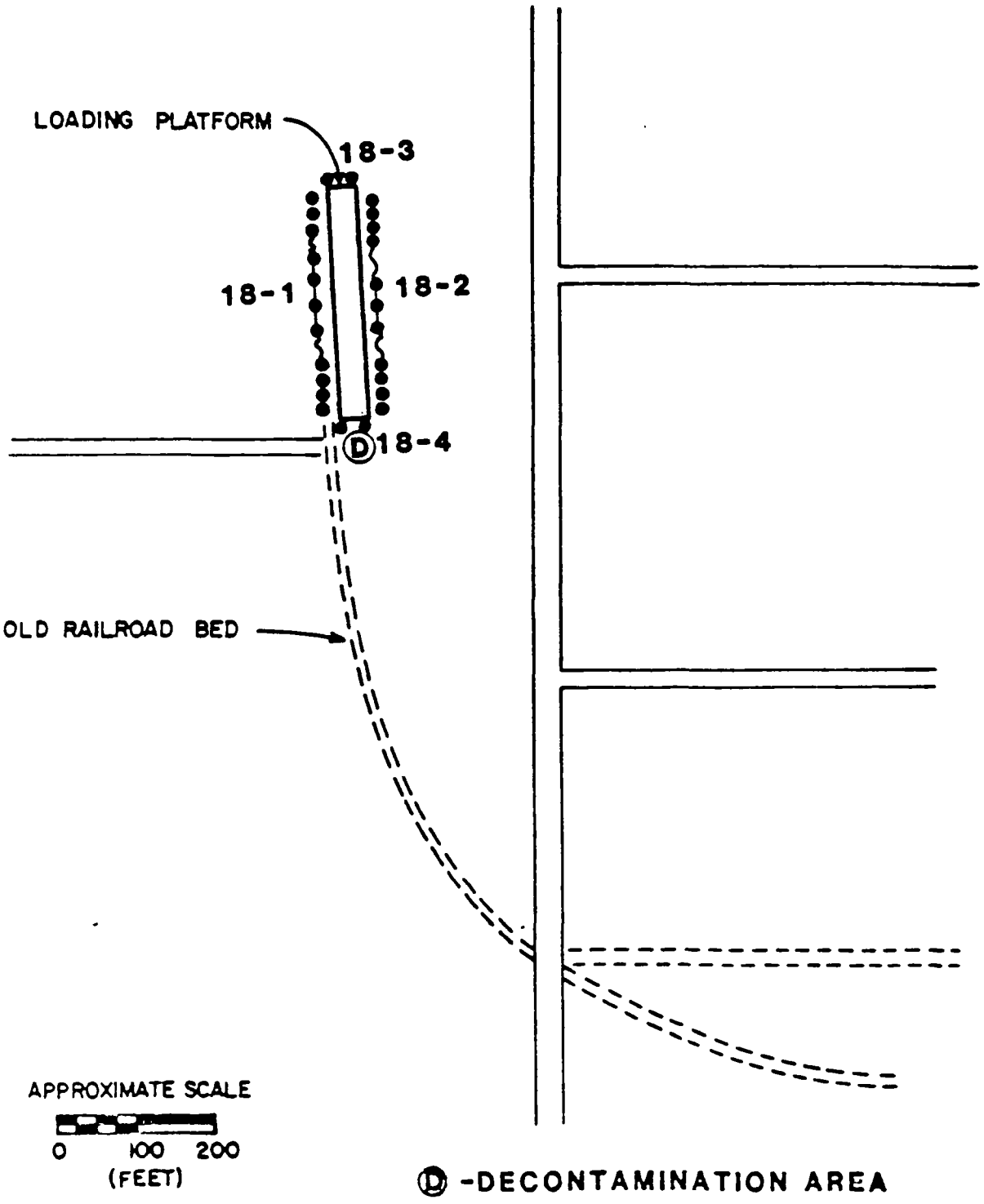
25.2 Site Investigations

25.2.1 Phase I Site Investigations:

Four composite soil samples (0-1 ft depth) were collected around the perimeter of the dock. Samples were collected from each of the two sides of the dock and from each end. The composites along the north and south consisted of 20 grabs, while the east and west composites consisted of 2 grabs each. Location 18-4, at the west end of the dock, was resampled for full priority pollutant analysis because it contained the highest FID scan reading at Site 18.

FIGURE 25-1

SITE 18
AREA 13 LOADING PLATFORM
PHASE I



O'BRIEN & GERE

25.2.2 Phase II Site Investigations:

No sampling was conducted in the Phase II investigation.

25.3 Analytical Results (See Appendix I, Page 18)

Trace quantities of the explosive tetryl were observed in two soil samples (1.90 mg/kg in each) from the north and east sides of the platform. Metals, volatiles and indicator parameter concentrations were similar to concentrations found in soils from the control sites. Two exceptions were noted: magnesium, 91,100 mg/kg in sample 18-4 and sodium, 2,330 mg/kg in sample 18-1, although these are estimated values only. CLP HSL organics analyses on sample 18-4 showed the presence of 4,050 ug/kg wet weight di-n-octyl phthalate. Several other semi-volatile organics were also detected at concentrations less than 300 ug/kg. Acetone and methylene chloride were detected, due to contaminants in the laboratory QA/QC blank. The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present.

25.4 Environmental Effects

25.4.1 Qualitative Assessment

This site was chosen for investigation based on its history of use as a loading dock for chemicals and explosives used in munitions manufacturing. It has also been reported that chemicals have been dumped on the site. The use of nearby areas as agricultural fields also warrants the need for an accurate site characterization.

Phase I analyses detected traces of the explosive tetryl. All other parameters were similar to the concentrations found in soils from the

control sites with the exception of magnesium and sodium in two different samples. However, at these concentrations, these metals will not threaten wildlife or affect human health. Di-n-octyl phthalate was also found in one sample, but it too was at a concentration below the Refuge background level.

Because there is no established waste source at this location, it is not possible to have a "complete" exposure scenario. Therefore, on the basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

25.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

25.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a site inspection and a sample analysis. The only unusual observation was a rusted drum shell located on a pile of dirt and stone rubble, indicating that some supplies may have been dumped when the loading platform was in use.

Chemical residue information consisted of analytical results for surface soil samples. This information was obtained only for the top one foot of soil; deeper soil borings were not conducted. Since there is no evidence to suggest that the surrounding soil has been disturbed, and the loading and unloading activities practiced at the site would likely

contribute only to surface contamination, these samples should adequately represent the site conditions.

It can be concluded that the data generated are adequate for evaluation of the remedial alternatives for this site. The analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

25.5 Preliminary Remedial Alternatives

Preliminary Phase I screening results discussed in the previous section indicated that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore this site was not included in the Phase II investigations. There will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

25.6 Conclusions and Recommendations

It can be concluded that the Area 13 Loading Platform does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

SECTION 26 - SITE 19, AREA 13 BUNKER 1-3

26.1 Site Description

Further information on Area 13 can be found in Section 25.1. It has been reported that chemicals may have been released near Site 19, Bunker No. 1-3, probably in the adjacent field. There is no observable impact on vegetation in the field except for one area of discolored vegetation. Wildspread presence of scattered red bricks suggests dumping has occurred at the site. An L-shaped area of brown vegetation was noted to the west side of one of the bunkers.

26.2 Site Investigations

26.2.1 Phase I Site Investigations:

Four composite soil samples (0-1 ft depth) were collected, one from each side of the bunker at distances up to about 125 feet. (See Figure 26-1). An additional composite soil sample was taken from the brown vegetation area.

26.2.2 Phase II Site Investigations:

One Phase I soil location (0-1 ft depth) was resampled for mercury analysis.

26.3 Analytical Results (See Appendix I, Page 19)

26.3.1 Phase I Analytical Results:

The FID scans showed low organic levels, on the order of 286-1,901 ug/kg, in the soils. The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support

FIGURE 26-1

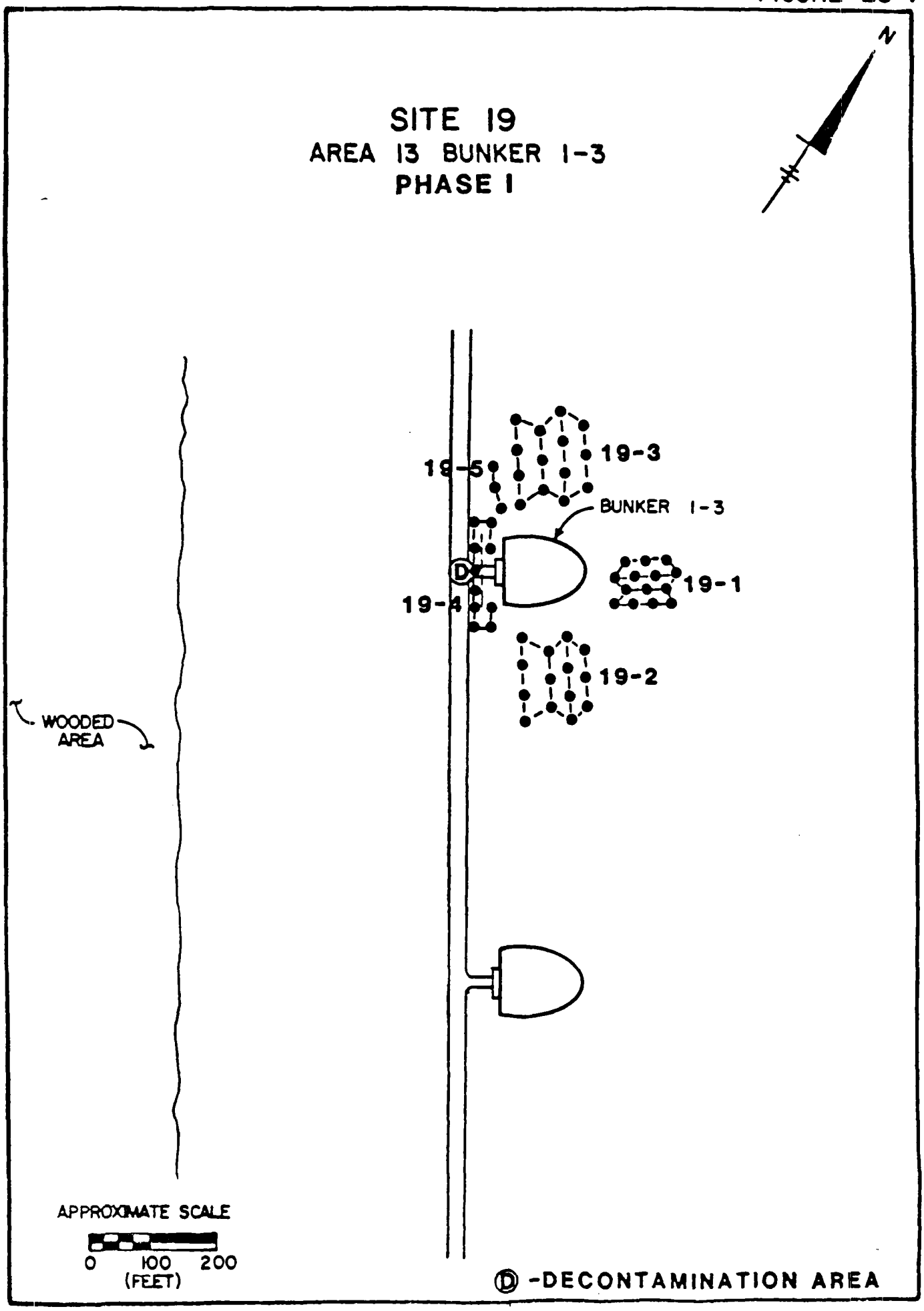
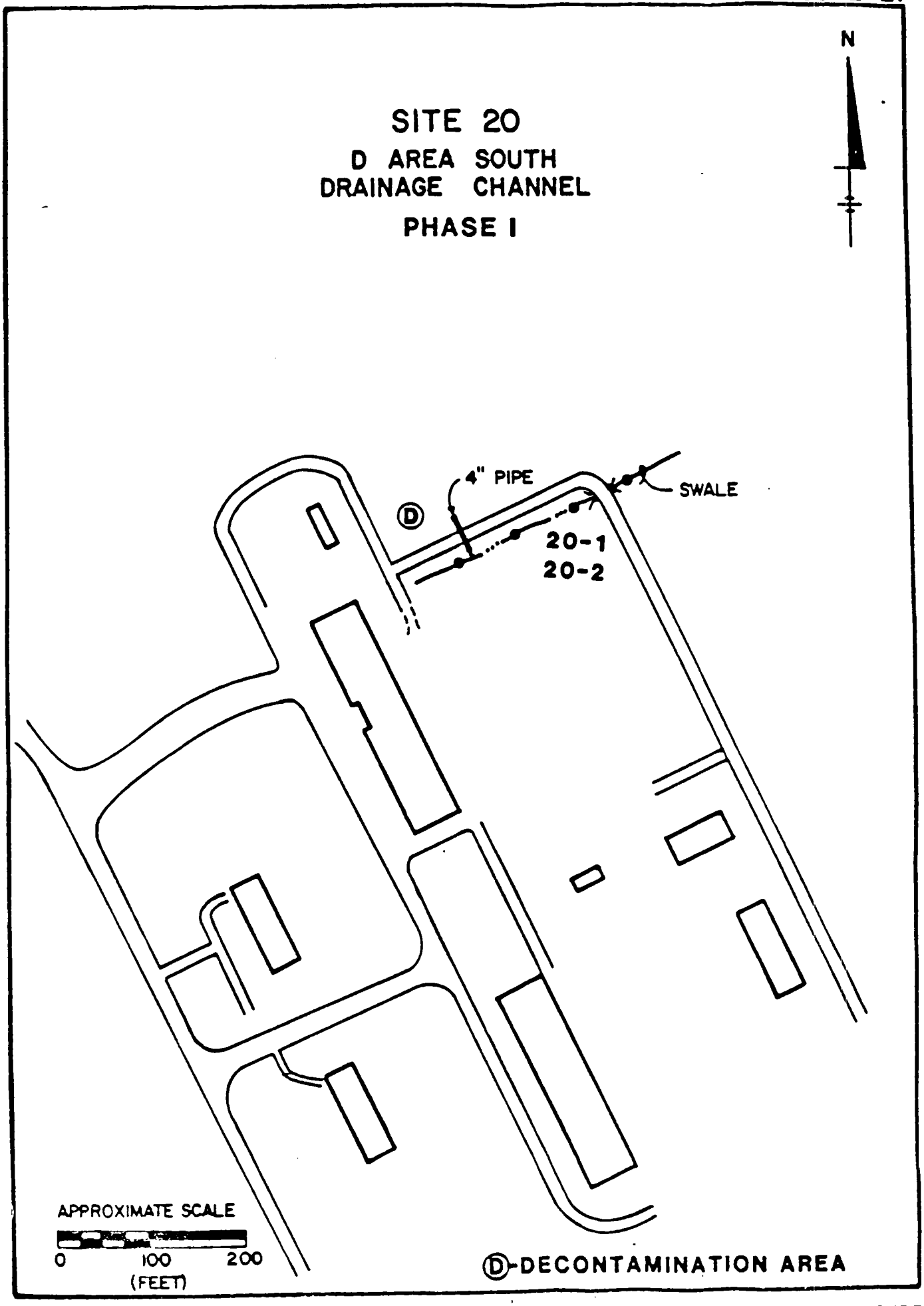

SITE 19
AREA 13 BUNKER 1-3
PHASE I
N
19-3
19-5
BUNKER 1-3
D
19-1
19-4
19-2
WOODED AREA
APPROXIMATE SCALE
0 100 200
(FEET)
D -DECONTAMINATION AREA

FIGURE 27-1



O'BRIEN & GERE ENGINEERS INC

data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present. The sample analyzed for full organics contained, on a wet weight basis, PCBs (1.1 mg/kg) and N-nitrosodimethylamine (1,455 ug/kg). The sample collected from the area of brown vegetation (location 19-5) contained 0.90 mg/kg of the explosive tetryl. Mercury was detected in one sample (19-4), from the front of the bunker, at 3 ug/kg, but this analysis was repeated in Phase II due to poor calibration data. All concentrations for metals in soils were similar to those detected at the control sites.

26.3.2 Phase II Analytical Results:

The mercury concentration in the second sampling of location 19-4 was 28 ug/kg, slightly above the detection limit of 20 ug/kg.

26.4 Environmental Effects

26.4.1 Qualitative Assessment

This site was chosen for investigation based on reports that chemicals had been poured onto the ground in an adjacent field. This dumping may have contaminated the surrounding area, as evidenced by a patch of discolored vegetation and an L-shaped area of brown vegetation.

Phase I analyses detected traces of the explosive tetryl and N-nitrosodimethylamine. N-nitrosodimethylamine has been shown to be carcinogenic in a number tests and is considered a suspect human carcinogen on this basis. The concentration of this compound at this site is higher than detected anywhere else on the Refuge, N-nitrosodimethylamine will be the focus of the risk assessment. The

Phase II mercury concentration was slightly above the detection limit but is within the level typically found in soil matrices and is not considered to pose a threat to wildlife or to human health.

26.4.2 Quantitative Assessment

The preliminary data from the Phase I survey indicate that N-nitrosodimethylamine is the contaminant of primary concern at this site, with 1,455 ug/kg detected in a site soil sample. Due to analytical deficiencies noted in the Phase I result for the single soil analyzed for this compound, and in order to ensure a conservative outcome from the analysis, the assessment below is based on twice the concentration detected, or 2,910 ug/kg. Because this compound has been shown to be carcinogenic to animals, a preliminary quantitative risk assessment will be performed, even though the data available are too limited to place a great deal of confidence in the result. Additional areas of uncertainty in the risk assessment are discussed in Section 26.4.3, Analysis of Uncertainties.

It is assumed that, during an excursion through this site, a human visitor to the site may ingest an average of 100 mg of site soil containing 2,910 ug/kg of N-nitrosodimethylamine as a result of direct contact exposure. This contact would result in an average exposure of 0.291 ug of N-nitrosodimethylamine per visit. Given the remoteness of the site and the absence of daily activities by humans in the area (the general area of the bunker sites is restricted by a locked fence to all but authorized personnel), visits to the site by humans do not occur on a daily basis. As a reasonable upper case exposure estimate, it is assumed that a Refuge employee visits the site once per month over a 30-year period, or 360 exposure days. U.S. EPA (1987) has estimated that a daily intake of

0.0137 ug of this compound each day for lifetime may be associated with an upper bound acceptable carcinogenic risk level of 10^{-5}. Utilizing the estimated intake of 0.291 ug/visit, an exposure of 0.004 ug/day is obtained when normalized over a 70 year lifetime. The estimated incremental cancer risk associated with this exposure level is 2.9×10^{-6}. This estimate, developed under a set of reasonable worst case exposure assumptions, is within the range of 10^{-4} to 10^{-7} risk considered acceptable to regulatory agencies for exposed populations.

The detection of N-nitrosodimethylamine in a soil sample analyzed in the Phase I survey of this site also presents a mechanism for exposure for terrestrial wildlife via the direct contact route. The levels of exposure would be greatest amongst small mammals as a result of inadvertent ingestion of contaminated soil residues and dust during daily burrowing, feeding and grooming. Thus, the risks of direct contact of these species to site nitrosamines residues are assessed. The risks to larger and/or less sensitive species, or to those which have less contact with soil residues would be proportionately lower. A search of on-line data bases (Pollution Abstracts, Biosis Previews, NTIS, HSDB) did not identify published studies on the effects of N-nitrosodimethylamine on pertinent wildlife species. Therefore, tests with surrogate species (i.e. laboratory rodents) are used in the assessment below.

Using the exposure assumptions detailed in Section 24.4.2.2 for exposures of burrowing mice, exposure to soil residues of 2.91 mg/kg N-nitrosodimethylamine will produce a daily intake of 2.76×10^{-2} mg/kg/day by the ingestion and inhalation routes. The most widely reported effect of chronic exposure of laboratory rats and mice to N-nitrosodimethylamine is the induction of hepatocellular carcinoma in a

number of investigations (HSDB, 1987). Using these data, U.S. EPA (1986) derived a unit risk factor of 26 $(mg/kg/day)^{-1}$. This value is an upper bound estimate of the slope of the tumor-exposure relationship, chosen as a conservative estimate of human response to carcinogen exposure at low concentrations. Assuming a similar sensitivity for wild rodent species, a cancer risk estimate of 0.72 is derived for burrowing rodents at this site. The significance of this risk level is uncertain. As discussed by Newell et al.(1987), concerns regarding the effects of cancer on wild populations are largely unknown, and risk levels of concern to humans are not directly transferable to wildlife. Many other factors come into play when addressing whether a wildlife population can maintain itself (i.e. survival to reproductive age, competition, weather, disease, predation, etc), and the effect of cancer, generally forming later in an exposed organism's lifetime, might thus be very small. On this basis, Newell et al. (1987) chose a risk level of 10^{-2} as a level of acceptable carcinogenic risk for wildlife, with the acknowledgement that more study is needed to justify this choice. Using this rationale, it is concluded that wildlife exposure to site residues of N-nitrosodimethylamine could result in a carcinogenic response, but the biological significance of the response cannot be assessed with currently available information.

Additional review of the literature on the effects of N-nitrosodimethylamine to wildlife is presented in USEPA's study on the Environmental Effects Profile on Nitrosamines (1986--). The authors used a reproductive effects study in mice to establish a minimum effective dose for N-nitrosodimethylamine. In this study, female mice were exposed to 0.1 mg/L of N-nitrosodimethylamine in drinking water for 75 days prior to mating, through pregnancy and weaning. Such exposure resulted in

significantly elevated fetal mortality in the treated group. Since mice consume approximately 5.7 ml of water per day, and the adult mouse body weight is approximately 30 g, an approximate exposure rate of 0.019 mg/kg/day is obtained as a lowest observed effect level (LOEL). The estimated daily exposure rate of 0.0276 mg/kg/day using twice the residue level detected at this site is roughly equal to this LOEL. Thus, assuming exposed species are of equivalent sensitivity to N-nitrosodimethylamine as laboratory mouse strains, and that the presence of this compound is widespread through the site at twice the level quantified in the survey, the estimated exposure rates may elicit a toxic response under subchronic conditions.

26.4.3 Analysis of Uncertainties

The information relied upon for evaluating this location consisted of a site inspection and sample analyses. The inspection of the site revealed one area of discolored vegetation and some evidence of dumping. Chemical residue information consisted of analytical results on surface soil samples. This information was obtained only for the top one foot of soil, whereas deeper soil borings were not conducted. Since there is no evidence to suggest that the surrounding soil has been disturbed, and the storage activities at this site would most likely not contribute to subsurface contamination, these samples should adequately represent the conditions of the site.

The quantitative risk assessment was performed under a worst case chronic exposure scenario of repeated monthly exposures and considered only the single data point available for a soil sample in which N-nitrosodimethylamine was detected in the Phase I survey. This result is

only qualitatively reliable due to insufficient QA/QC supporting the analysis. In addition, the residue concentration detected was doubled in the risk calculation to provide a more conservative outcome. This worst case approach resulted in a risk estimate on the lower end of the 10^{-6} to 10^{-4} level generally considered as an acceptable range of risk to humans. The exposure level estimated for inherent populations of small burrowing mammals could measurably affect reproduction of such species if these were to meet the set of worst case assumptions used in the assessment. However, the actual level of risk posed by site-related contaminant exposure cannot be stated with confidence without a more thorough sampling of the site.

26.5 Preliminary Remedial Alternatives

The [illegible]al results and the evaluation of environmental effects for this site indicated that N-nitrosodimethylamine residue levels in soil may pose unacceptable exposure levels to small wild rodent species. The risks to potential human receptors were determined to be on the order of 10^{-6}, which is a level generally considered acceptable. Due to previous activities in this area, including use of the bunker sites for storage of munitions, explosives, and other wartime supplies, the nitrosamines residues in a soil sample from this site may be the result of degradation of chemicals previously stored in this area. No further evaluation of remedial alternatives will be conducted for this site; however, it is recommended that the Refuge Management initiate additional investigations in this area to determine if further action will be necessary to protect potentially exposed wildlife.

26.6 Conclusions and Recommendations

It can be concluded that the Area 13 Bunker 1-3 site does not represent a risk of exposure to human health, but may be affecting resident wildlife due to the presence of low level N-nitrosodimethylamine residues in soil. It is recommended that additional investigations be initiated to further evaluate the potential risks associated with residues at this site. This site will not be evaluated further as part of this RI/FS.

SECTION 27 - SITE 20, D AREA SOUTH DRAINAGE CHANNEL

27.1 Site Description

Area D is an active Olin operation located north of Crab Orchard Lake. This area is currently used for the manufacture of explosives. The site was previously used by Universal Match under contract to the DOD. Their operations ceased after a large explosion, according to the Refuge Manager.

An abandoned building is located within the fenced southeastern end of the Olin D Complex. It was reported that chemicals were dumped here. Site 20 consists of a drainage swale originating at the building that runs east outside of the fence. (See Figure 27-1). A four-inch pipe (dripping at the time of the site inspection) extends from the Olin Area under the fence and discharges to this ditch. A slight sheen was noticeable on the surface water in pooled areas of the ditch.

27.2 Site Investigations

27.2.1 Phase I Site Investigations:

One sediment composite of four grab samples (0-1 ft depth) was collected. The sediment was resampled for full organics analysis. One water sample was scheduled but could not be collected because the ditch was dry.

27.2.2 Phase II Site Investigations:

Cyanide and mercury analyses were scheduled for one water sample from the ditch; however, the ditch was dry at the time of sampling and the sample could not be collected.

27.3 Analytical Results (See Appendix I, Page 11)

Cyanide (13 mg/kg) and mercury (8.9 ug/kg) were detected in the sediment composite, however, these data are questionable due to QA/QC deficiencies. The sediment was analyzed for full CLP organics after an FID screen of 16,477 ug/kg; it contained 30,500 ug/kg wet weight di-n-octyl phthalate, 2320 ug/kg wet weight bis (2-ethylhexyl) phthalate, and 336 ug/kg wet weight N-nitrosodimethylamine. All other organics were below the detection level or were reported but were also present in the laboratory QA/QC blank. The volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present.

27.4 Environmental Effects

Envi[illegible]f drainage within the D and P areas are discussed in Section 16.[illegible].

27.5 Preliminary Remedial Alternatives

Preliminary Remedial Alternatives for the D and P areas are discussed in Section 16.5.

27.6 Conclusions and Recommendations

Conclusions and Recommendations for the D and P areas are discussed in Section 16.6.

SECTION 28 - SITE 21, SOUTHEAST CORNER FIELD

28.1 Site Description

Site 21 is a fenced field (150 ft. x 400 ft.) located at the southeast corner of the Refuge. (See Figure 28-1). The field is thought to be the site of an old dump due to the presence of concrete rubble near one end. No other evidence of debris is observable. The topography gradually slopes to the south and east toward a swampy ditch at the bottom of the slope. Large diameter trees in the field indicate the area has not been disturbed for at least sixty to seventy years.

28.2 Site Investigations

28.2.1 Phase I Site Investigations:

A magnetometer and electromagnetic terrain conductivity survey was conducted along four north-south transects. (See Figures 28-2 and 28-3). Four composite soil samples (0-1 ft depth) were collected, one along each transect. One composite along transect 1 was resampled for full organics analysis.

28.2.2 Phase II Site Investigations:

The transect 1 soil composite was resampled and analyzed for mercury.

FIGURE 28-1

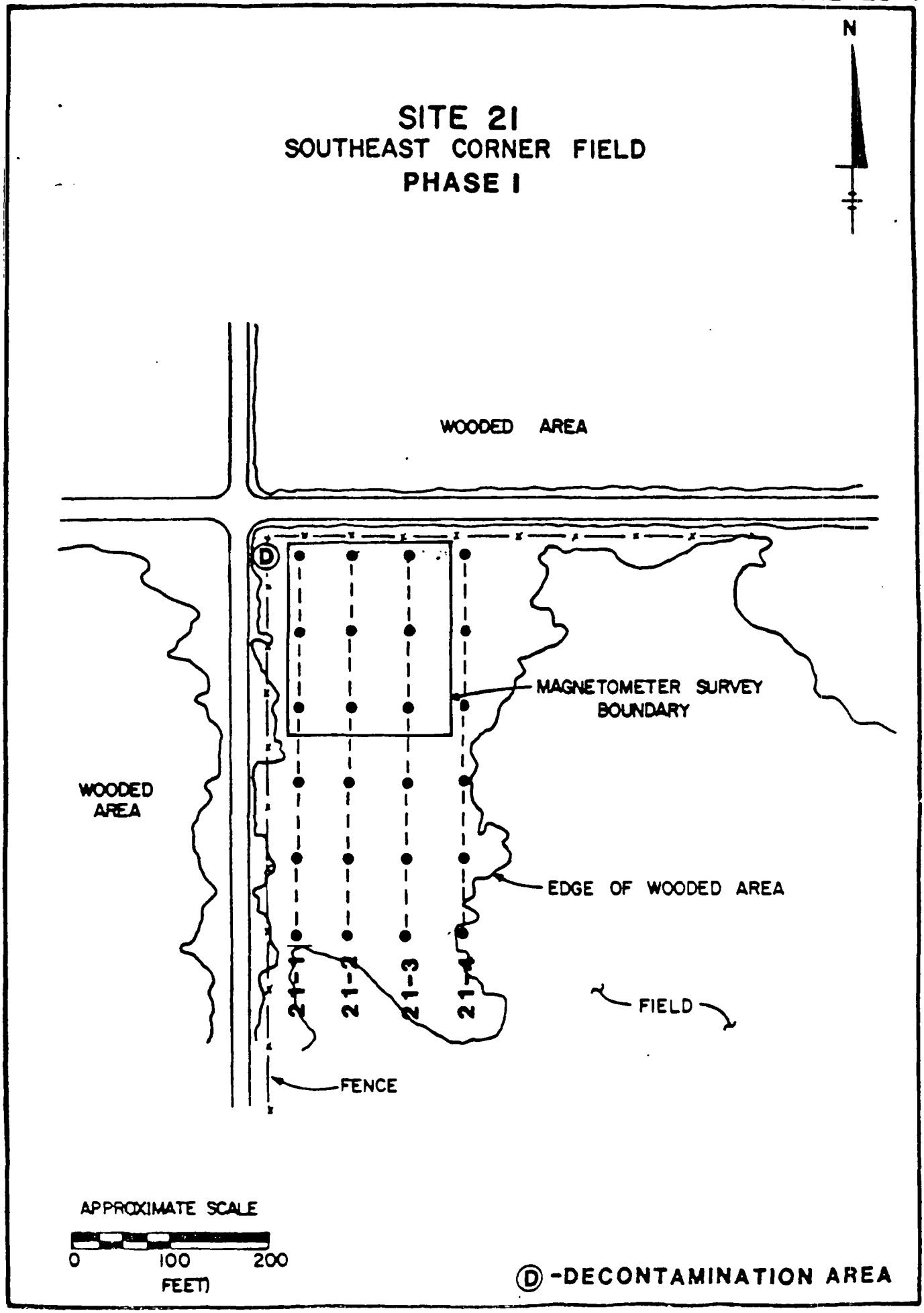

N
SITE 21
SOUTHEAST CORNER FIELD
PHASE I
WOODED AREA
WOODED AREA
D
MAGNETOMETER SURVEY BOUNDARY
EDGE OF WOODED AREA
21-1
21-2
21-3
21-4
FIELD
FENCE
APPROXIMATE SCALE
0
100
200
FEET)
D -DECONTAMINATION AREA

FIGURE 28-2

SITE 21

MAGNETOMETER SURVEY

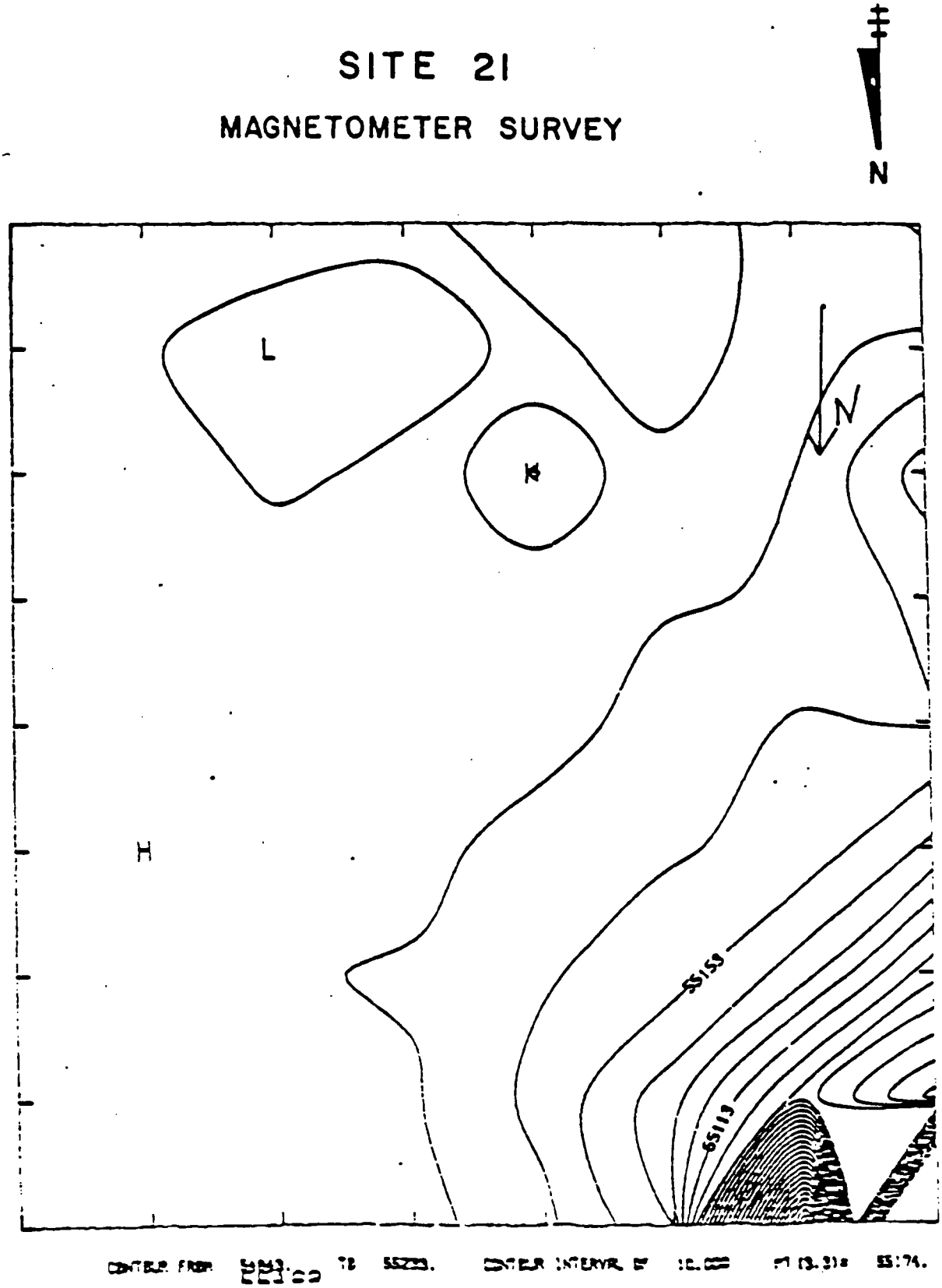
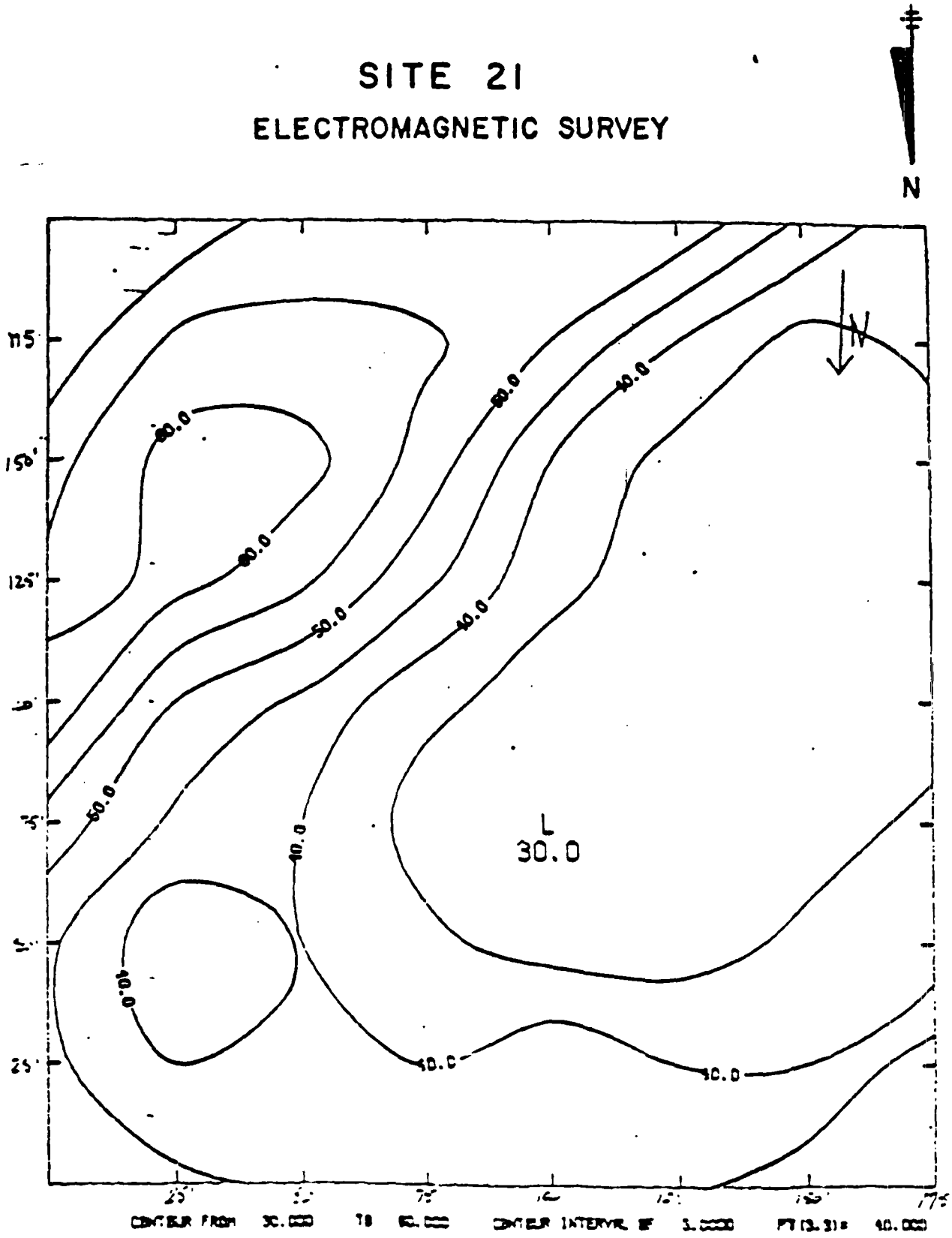


FIGURE 28-3

SITE 21

ELECTROMAGNETIC SURVEY



O'BRIEN & GERE

28.3 Analytical Results (See Appendix I, Page 22)

28.3.1 Phase I Analytical Results:

The magnetometer and electromagnetic survey as shown in Figures 28-2 and 28-3 did not indicate any unusual subsurface conditions.

Two of the soil samples contained high magnesium levels (10,500 and 27,200 mg/kg, wet weight), which were approximately one order of magnitude higher than the levels detected at the control sites (metals are estimated values only). The FID scan on the composite soil sample from transect 1 was 20,630 ug/kg (25,274 ug/kg duplicate); however, only trace base/neutral extractable compounds were detected. The semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present. N-nitrosodiphenylamine (156 ug/kg wet weight) was the only compound detected above the detection limit, although traces of other semi-volatiles were also reported. The sample representing transect 1 also contained 9 ug/kg of mercury; all other soil samples contained mercury below the 1 ug/kg detection limit. The mercury analysis was repeated in Phase II due to poor calibration data.

28.3.2 Phase II Analytical Results:

The soil composite contained 41 ug/kg of mercury.

28.4 Environmental Effects

28.4.1 Qualitative Assessment

This site was chosen for investigation based on the thought that it had been an old landfill at one time. The site also slopes towards a swampy drainage ditch, which would be a viable transport mechanism, should the site be contaminated.

Phase I sampling analysis detected traces of magnesium, but the concentration was below the detection limit. N-nitrosodiphenylamine was detected but at concentrations over ten times below those detected at other sites where this compound was not considered to represent a significant risk of exposure (see Section 19.4). One sample also contained traces of mercury and a Phase II soil sample was taken to more accurately quantify this result. The Phase II analysis showed a slightly higher concentration than the Phase I result, but is not considered to be detrimental to the environment.

Because there is no established waste source at this location, it is not possible to have a "complete" exposure scenario. Therefore, on the basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

28.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

28.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a site inspection, geophysical surveys, and sample analyses. An inspection of the site led to the belief that the site had been an old dump. However, the geophysical surveys did not reveal any unusual subsurface conditions. Large diameter trees in the field indicated that the area had not been disturbed for at least sixty to seventy years.

Chemical residue information consisted of analytical results on surface soil samples. This information was obtained only for the top one foot of soil; deeper soil borings were not conducted. Based on the magnetometer and electromagnetic terrain conductivity surveys, there is no evidence to suggest that waste had been buried on this site. Therefore, these samples should adequately represent the conditions of the site.

It can be concluded that the data generated are adequate for evaluation of the remedial alternatives for this site. The sampling analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

28.5 Preliminary Remedial Alternatives

The analytical results discussed in the previous section indicate that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore there will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

28.6 Conclusions and Recommendations

It can be concluded that the Southeast Corner Field site does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

SECTION 29 - SITE 22, OLD REFUGE SHOP

29.1 Site Description

North of the Refuge along Wolf Creek Road is the old Refuge Headquarters, now leased by Diagraph-Bradley. Site 22, the Refuge Shop, was located behind the Headquarters building. Pine wood poles were treated in a fenced area of the Shop with pentachlorophenol wood preservative and shipped to various locations throughout the county, according to the Refuge Manager. A small drainage pool is located outside the fence to the north and contains a green-yellow scum. (See Figure 29-1). The pool drains through the woods to the northwest and ultimately into Crab Orchard Lake.

29.2 Site Investigations:

29.2.1 Phase I Site Investigations:

One grab surface water sampled from the drainage pool. One composite sediment sample (0-1 ft. depth) was collected from the drainage ditch. The sediment was resampled for full priority pollutant analyses.

29.2.2 Phase II Site Investigations:

A monitoring well was installed and sampled during Phase II. The monitoring well was set to a total depth of 10 feet in silty clay and was screened from 5 to 10 feet. The ground water was sampled and analyzed for CLP HSL volatiles, base/neutral/ acid extractables, and metals.

FIGURE 29-1

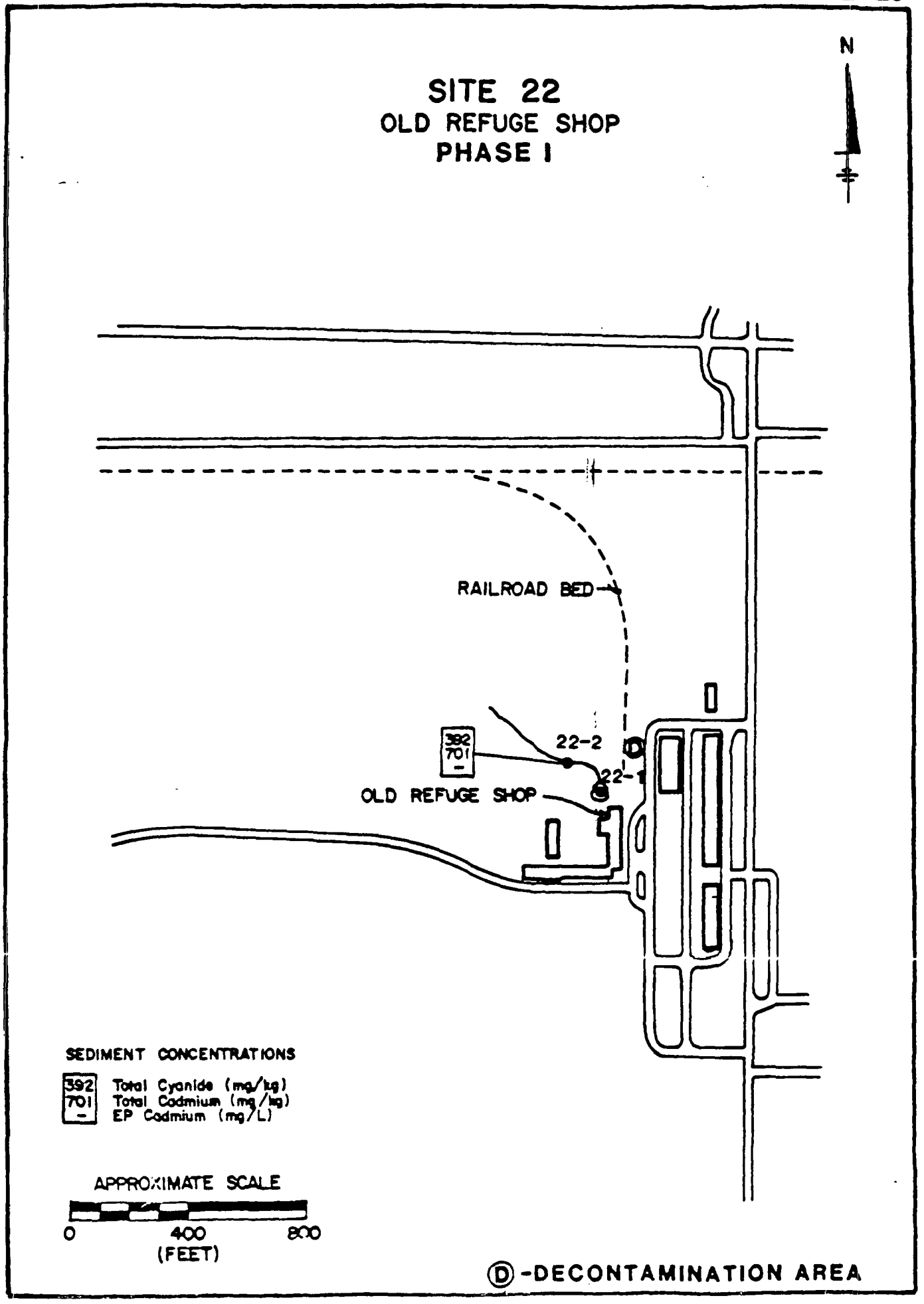
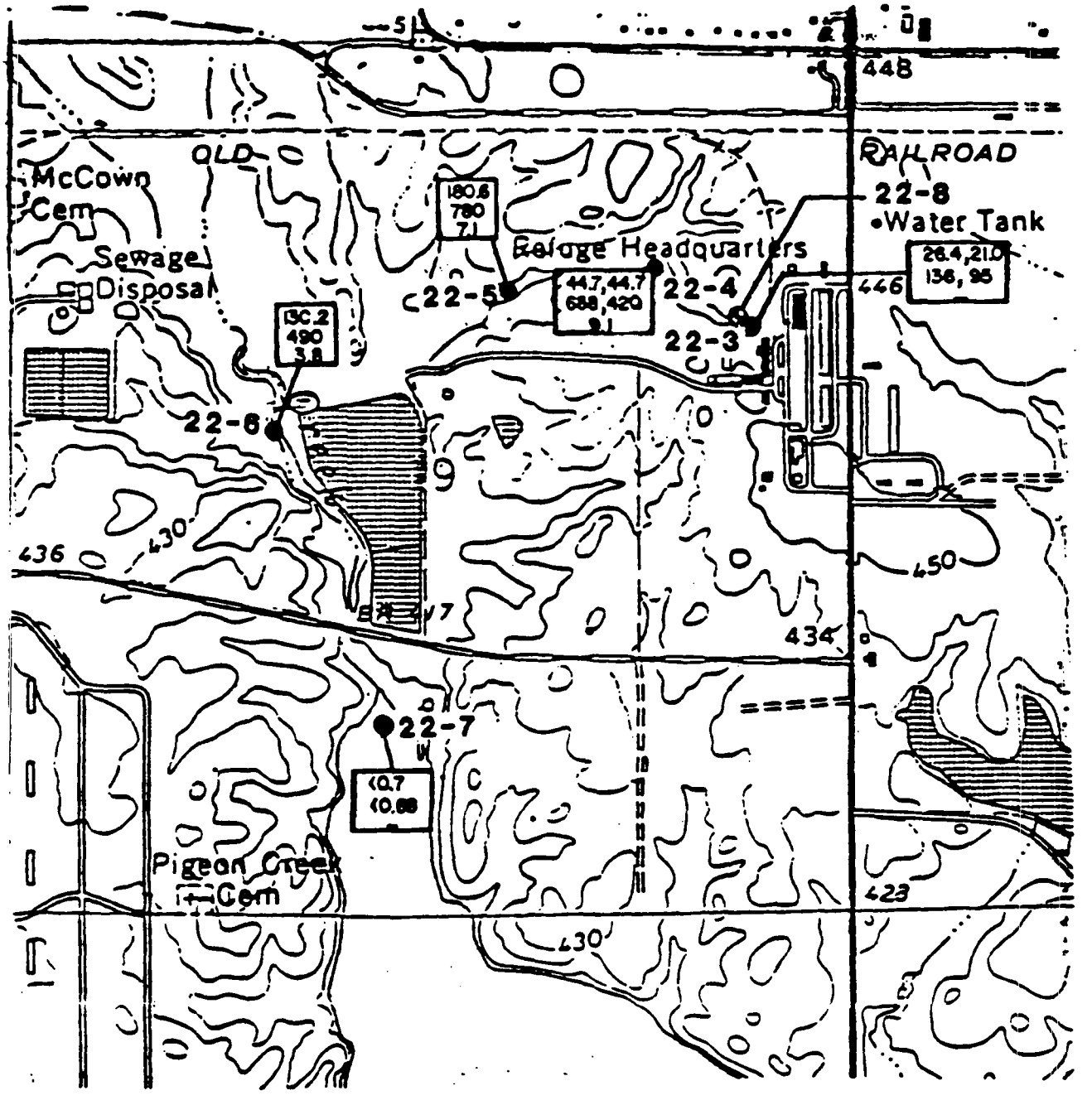

SITE 22
OLD REFUGE SHOP
PHASE I
N
RAILROAD BED
392
701
-
22-2
22-1
D
OLD REFUGE SHOP
SEDIMENT CONCENTRATIONS
392 Total Cyanide (mg/kg)
701 Total Cadmium (mg/kg)
- EP Cadmium (mg/L)
APPROXIMATE SCALE
0
400
800
(FEET)
Ⓓ-DECONTAMINATION AREA

FIGURE 29-2

SITE 22
SAMPLING LOCATIONS
PHASE II



SEDIMENT CONCENTRATIONS

130.2	Total Cyanide (mg/kg)
490	Total Cadmium (mg/kg)
3.8	EP Cadmium (mg/L)

⊕ Shallow well

SCALE IN FEET

0 1000 2000

N

Four sediment samples were collected from the ditch. One additional soil sample was collected from the embankment to trace the downstream distribution of contaminants (see Figure 29-2). The soil and sediment samples were analyzed for CLP base/neutral/acid extractables, as well as cadmium, chromium, and cyanide. EP-toxicity analyses were completed on three of the sediment samples.

29.2.3 Site Hydrogeologic Characterization

29.2.3.1 Site Geology

Based on results of the test well boring 22-8, the subsurface unconsolidated overburden consists of a brown, gray, and orange mottled silty clay, with some sand. This material is present from the ground surface to 9 ft. in depth. Beneath the silty clay, at least 1 ft of a brown silt with a trace of fine gravel is present to 10 ft in depth (total depth of boring). Bedrock was not encountered in the boring; therefore, the depth to bedrock and bedrock lithology is unknown. As only the one monitoring well was installed, the lateral extent and variability of the overburden units is also unknown.

29.2.3.2 Site Hydrogeology

Shallow ground water occurring beneath the site was found at a depth of 1 to 1.3 ft. below the ground surface within the silty clay soil unit during June 1987. The monitoring well installed screened this upper water table. Ground water elevations collected during the winter and summer of 1987 (wet and dry seasons, respectively) indicate a water table fluctuation of 0.3 ft with water levels dropping during the summer

months (Table 4-3). Figure 35-5 illustrates the monitoring well location and the ground water elevation of 18 June, 1987.

29.3 Analytical Results (See Appendix 1, Page 22)

29.3.1 Phase I Analytical Data:

The pool water sample did not contain contaminant concentrations above the Illinois Public Water Supply Standards or Federal drinking water standards. The total organic halides concentration in water was 14 ug/L (16 ug/L duplicate). The sediment contained cyanide (392 mg/kg), cadmium (701 mg/kg), chromium (663 mg/kg), and lead (150 mg/kg) above the concentrations detected at the control sites. The metals concentrations are reported as estimated values and cyanide analyses were repeated due to QA/QC deficiencies (see Exhibit B). All organics were below detection limits, although the FID scan was 10,114 ug/kg. Methylene chloride and acetone were quantified, but these were also detected in the laboratory blank. The total organic carbon concentration in the sediment was 19,413 mg/kg, and organic nitrogen was 1,899 mg/kg.

29.3.2 Phase II Analytical Data:

The ground water sample contained low levels of cadmium (25 ug/L), chromium (21 ug/L), and lead (6.6 ug/L). The corresponding dissolved metals concentrations were 17, 5.1, and 3.9 ug/L. Cyanide was detected at 0.07 mg/L. No organics were detected in the HSL/CLP analysis, although the spike/spike duplicate recoveries were outside of control limits and the semi-volatiles were extracted outside of the holding time. All parameters were within the Illinois Public Water Supply

Standards, except for cadmium and cyanide. The cadmium concentration also exceeded the Federal drinking water MCL and MCLG.

Sediment samples 22-5 and 22-6 downstream contained the highest cyanide concentrations, 130 mg/kg and 181 mg/kg wet weight. The sediments also contained cadmium (less than 0.68 - 780 mg/kg), chromium (10-889 mg/kg) and lead (93 - 166 mg/kg). Chromium duplicate analyses were not within the control limits. In general, the sediments which contained high cyanide levels also contained high metals concentrations. Three sediment samples, extending approximately 3000 ft. downstream of the pool, contained EP Toxic cadmium concentrations (9.1, 7.1, and 3.8 mg/L) in excess of the RCRA criterion of 1.0 mg/L, thus defining these sediments as hazardous wastes. Figure 29-2 shows the cyanide and total and extractable cadmium concentrations in the sediments. Trace base/neutral/acid extractable compounds were found in soil 22-3 (and duplicate 40-76), including 2-methylnapthlalene (330 and 280 ug/kg), bis(2-ethylhexyl) phthalate (200 and 320 ug/kg), and di-n-butyl phthalate (1,260 and 499 ug/kg), although the recoveries for spike and spike duplicate samples were outside of limits for the semi-volatiles scan. Other detected organics were also present in the QA/QC blanks.

29.4 Environmental Effects

29.4.1 Qualitative Assessment

29.4.1.1 Source Evaluation

The results of the site investigations, as described in the preceding sections, determined that the vicinity of the Old Refuge Shop (previously used to treat wood poles with preservative) was contaminated

with residues of cadmium, cyanide, and other compounds. Phase I analyses of water and sediments in a small drainage ditch adjacent to the shop showed the presence of 701 mg/kg cadmium, 603 mg/kg chromium, and 392 mg/kg total cyanide in ditch sediments while the water contaminant levels were less than the Illinois Water Supply Standards. A shallow well installed on the site showed only very low levels of contaminants, but cadmium and cyanide levels exceeded the State and Federal standards. Phase II soil and sediment EP Toxicity analyses at the site and 3000 feet downstream confirmed the site contamination and offsite transport of cadmium. Based on this analytical survey, cadmium and cyanide were chosen to serve as site indicator contaminants for the purpose of this risk assessment.

The physicochemical and toxicological properties of cadmium and cyanide are summarized in Exhibit A. Cadmium is a highly toxic element capable of producing a broad range of systemic effects, particularly to the respiratory, renal, and reproductive systems. It is also a teratogen in animal studies, and there is evidence for carcinogenicity in humans via inhalation as well. Cadmium can accumulate extensively in exposed individuals and in populations through food chain magnification of residues. The major concern from cyanide exposure is acute toxicity of hydrocyanic gas (HCN) and simple salts such as sodium cyanide. HCN would exist as a gas under normal environmental conditions, while the chemistry of the salts is quite complex and compound-specific, with a variety of soluble and insoluble complexes that can be formed. Cyanide does not bioaccumulate.

29.4.1.2 Transport Route Evaluation

a) Air: Cadmium has no propensity to volatilize to air from the adsorbed state, Therefore, transport of cadmium compounds in vapor form is not a significant route of transport. However, because of the existence of exposed soil-adsorbed contaminants, dusts generated by wind erosion, vehicular traffic, or the activities of endemic wildlife constitute a functional route for conveying cadmium residues to on- and off-site locations for subsequent exposures by receptors in those areas. Due to the high vapor pressure of HCN and the relatively low volatility of ionic cyanide, exposures via both the vapor and dust-bound forms of cyanide are possible.

b) Direct Contact: Due to the presence of site indicator contaminants in soils and sediments in the area, exposures by the direct contact route are possible.

c) Surface Water: Phase II analyses detected cadmium residues in excess of 1 mg/kg in sediments 3000 feet downstream of the site drainage pool. Therefore, the surface water transport route is considered functional via precipitation-initiated runoff events which convey soil- and sediment-bound site contaminants towards Crab Orchard Lake.

d) Ground Water: Only minimal quantities of site contaminants were detected in site ground water, and cadmium mobility in silty clay soils is generally low. Furthermore, no exposed receptors (ground water users) were identified for this route. Therefore, the ground water transport route was determined to be non-functional at the site and will not be considered further in this risk assessment.

29.4.1.3 Receptor Evaluation

Human

The Old Refuge Shop is situated in a non-populated area. Therefore, the only potential human receptors would include facility employees, site trespassers, and occasional recreational users of the Refuge. The number of human receptors is low and exposure will be of a transient, non-chronic nature.

Specific scenarios for human exposure to site indicator contaminants will be developed in the following sections. The transport route evaluation identified three functional transport mechanisms: the air route, the direct contact route, and the surface water transport route. Exposures will generally occur only in the vicinity of the Shop and drainage ditch, with the exception of downgradient drainage conveying surface water towards the lake, and potentially from consumption of fish taken from the lake.

The following are the most likely human exposure scenarios for the functional transport routes.

a) Direct Contact: The most probable human exposure scenario would be exposure to site indicators via direct contact with surface residues and sediments at the site and drainage ditch and to sediments in the downstream drainage areas. Humans employed at the facility constitute one group of potential receptors. Occasional recreational users of the Refuge might also be exposed to contaminants via direct contact. The most likely mode of entry of contaminants into the body would be incidental ingestion of soil-bound residues adhering to the skin, clothing, or shoes acquired by direct contact with exposed wastes.

b) Air Route: Dusts generated by wind erosion or foot traffic over exposed waste areas constitute the most likely mechanism for exposure via the air route since cadmium and some forms of cyanide are soil-bound and non-volatile. HCN is highly volatile and thus poses a potential additional exposure mechanism if present at the site. As with the direct contact pathway, the receptors include facility employees and other trespassers who may breathe contaminated dusts while traversing the site.

c) Surface Water Route: As presented in the preceding section, transport of cadmium residues towards Crab Orchard Lake presents a potentially complete human exposure pathway via ingestion of residues accumulated in fish.

d) Ground Water Route: No human users of site ground water were identified. Therefore, this exposure pathway is incomplete.

Wildlife

The forested nature of the site and adjoining areas indicates that a wide variety of terrestrial organisms may be exposed to site-related contaminants, and the proximity to Crab Orchard Lake creates the opportunity for exposures to aquatic populations.

a) Direct Contact: Wildlife inhabiting the site such as invertebrates and small burrowing rodents will receive both acute and chronic direct contact exposures to site contaminants bound to soil dusts during burrowing activities. Exposed waste areas present a functional direct contact route exposure path for birds while feeding, ingesting grit, and dusting. Subsequent ingestion of soil-bound residues while preening or grooming is the principal means of entry into the body.

b) Air Route: Inhalation exposures of wildlife to dust-bound cadmium and cyanide will follow the direct contact scenario described above. In addition, inhalation of HCN vapor is possible.

c) Surface Water: Wildlife using the Shop drainage pool for drinking water will inadvertently ingest sediments containing residues of cadmium, chromium, and cyanide. In addition, transport of cadmium to Crab Orchard Lake via runoff of sediments is a potentially functional chronic exposure pathway for aquatic organisms in Crab Orchard Lake. Exposures will be relatively greatest for benthic invertebrates and bottom-feeding fishes such as catfish. It is noted, however, that cyanide and cadmium were not detected in the sediments of Crab Orchard Lake near the mouth of the stream leading from the Old Refuge Shop. These compounds were, likewise, not detected elsewhere in the sediments of Crab Orchard Lake.

d) Ingestion: Implied in all three wildlife exposure pathways discussed above is the ingestion of site contaminants via soils, dusts, sediments, vegetation, water, and consumed prey. In addition, herbivores may consume contaminated dusts on seeds and vegetation. Fish, birds (i.e. ducks, herons) and other aquatic organisms may inadvertently ingest contaminant-bearing sediments while feeding. The ability of cadmium to accumulate in aquatic and terrestrial food chains is well documented, adding to the importance of the ingestion route of exposure.

e) Ground Water: No surface ground water discharges have been shown which would provide a complete pathway for wildlife exposures.

29.4.2 Quantitative Assessment

29.4.2.1 Estimates of Release and Exposure Rates

Estimates of Exposures by Direct Contact

The qualitative assessment for the Old Refuge Shop has determined that direct contact represents a functional exposure pathway for humans and wildlife. However, cadmium and some forms of cyanide are tightly bound to soils and sediments. Therefore, dermal absorption of contaminants is not expected. The pathway consists, instead, of ingestion of bound residues picked up through direct contact with soils and sediments. The contribution of this route of exposure will therefore be addressed in the section below on ingestion exposures.

Estimates of Airborne Exposures

The qualitative portion of this assessment has established that the air pathway represents a complete exposure route. The pathway consists of breathing contaminated dusts at the site by occasional human activities (visits by employees, recreational users etc.), and by burrowing and dusting activities of wildlife.

The general approach and assumptions used to estimate airborne human and wildlife exposures are given in Section 24.4.2.1 of this report. Using this worst case approach for a four hour excursion by a facility employee or hiker in a sector of the site containing exposed wastes, and assuming a mean surface cadmium soil/sediment level of 500 mg/kg, a total exposure to cadmium of 0.026 mg or 0.37 ug/kg for a 70 kg adult is obtained via the inhalation route per site visit. Assuming three such visits to the site per year, a chronic inhalation rate of 0.003 ug/kg/day is derived. It should be realized that such a scenario does not technically define a chronic exposure. Using the same approach for

exposure to dust-bound cyanide at 100 mg/kg, an exposure of 0.074 ug/kg body weight per site visit is obtained. For repeated exposures, e.g. 3 times yearly, this intake would be expressed as 0.0006 ug/kg/day. The contribution of inhaled residues to total chronic intake is discussed in the following Section 29.4.2.2, Quantitative Assessment.

True chronic inhalation exposures are likely, however, for small burrowing mammals at the site. Assuming an average of 1 hr daily burrowing and breathing using a breathing rate value of 0.006 m3/hour for an active 30 g mouse (approximately U.S. EPA, 1985), four times the resting rate cited in small rodents might receive exposures up to 1.0 mg/kg/day as a result of burrowing in soils containing 500 mg/kg cadmium. Vaporized residues would not be significant due to the very low volability of the cadmium compounds. For 100 mg/kg cyanide, a daily exposure of 0.2 ug/kg/day is estimated. Due to lack of monitoring data, exposure to HCN gas cannot be estimated. Given the acute lethality of this substance, its presence in substantial amounts would be obvious. The significance of this exposure is discussed below in 29.4.2.2.

Additional wildlife species are considered in the following section on Quantitative Assessment.

Estimates of Ground Water Exposures

It has previously been determined that the groundwater exposure pathway is incomplete at the site and therefore will not be considered quantitatively.

Estimates of Surface Water Exposures

In view of a functional transport mechanism for conveying site contaminants towards Crab Orchard Lake via runoff events, the surface water pathway is complete. Exposures due to direct contact with sediments is discussed below under ingestion exposures. Another mechanism of exposure may consist of ingestion of contaminants accumulated in biota from residues present in sediments transported to Crab Orchard Lake. Therefore, exposures by this route will be discussed in the following section on ingestion exposure.

Estimates of Ingestion Exposures

Ingestion exposure of site contaminants at the Refuge Shop and contiguous sites has two components: ingestion of soil-bound residues acquired by direct contact with waste materials, and bioconcentration and foodchain accumulation of cadmium in terrestrial communities and possibly in Crab Orchard Lake. The approaches and assumptions used to estimate exposures by direct contact and ingestion of contaminated soils have been discussed in section 24.4.2, the quantitative assessment for the Job Corps site. Using the worst case assumption that an individual ingests 100 mg of soil as a result of an excursion into an exposed waste area of the site and that the mean surface level of cadmium and cyanide are 500 and 100 mg/kg, respectively, an ingestion of 0.71 ug/kg for cadmium and 0.14 Ug/kg for cyanide are estimated per site visit for a 70 kg human. The corresponding chronic exposure levels of 0.0058 and 0.0011 ug/kg/day could be derived if it is assumed that site visits by humans recur at least 3 times per year.

The following wildlife cadmium and cyanide intake rates from ingestion of contaminated soil at the site during feeding or grooming are estimated using similar assumptions as those detailed in Section 24.4.2 The estimated exposure from 500 mg/kg cadmium in soil are: rabbit, 7.1 mg/kg/day; mouse, 4.75 mg/kg/day; and deer, 2.69 mg/kg/day. For cyanide at 100 mg/kg soil, the corresponding estimates are: rabbit, 1.4 mg/kg/day; mouse, 0.95 mg/kg/day; and deer, 0.54 mg/kg/day.

Beyer et al (1982) determined that earthworms of the family Lumbricidae were capable of bioaccumulating cadmium directly from soil. For example, earthworms living in soil containing 2 mg/kg of cadmium contained levels of cadmium as high as 100 mg/kg body weight. The authors considered these cadmium levels to be hazardous to wildlife which might feed on the worms, although no experimental evidence for this quantitative conclusion was provided. Given the scarcity of data on the effects of ingested cadmium to carnivorous wildlife which might receive such exposures (i.e. shrews, moles, skunks, certain birds, for example), a comparison of possible exposure rates to established effect levels is made in Section 29.4.2.2, Quantitative Risk Assessment for wildlife.

29.4.2.2 Quantitative Risk Assessment

Human Risks

Human exposure at the Refuge Shop site will be limited, with little opportunity for true chronic exposures. Nevertheless, the assessment below carries the scenario through to potential chronic exposures to determine the level of risk to humans. Human exposure estimates for airborne dust-bound cadmium residues (0.003 ug/kg/day) and direct contact ingestion of soil-bound residues of cadmium (0.0058 ug/kg/day)

provide a total estimate of 0.009 ug/kg/day for this scenario. A unit risk factor of 7.8 $(mg/kg/day)^{-1}$ has been established by U.S. EPA (Exhibit A) for assessing human carcinogenicity based on evidence that inhaled cadmium has produced respiratory cancer in the workplace. Using this value and the estimated inhalation exposure rate of 0.003 ug/kg/day, an incremental risk of 2.3×10^{-5} is estimated. This upper limit estimate is very close to the 10^{-5} to 10^{-6} population risk considered by regulatory agencies to constitute a negligible risk to the national population. In view of the low probability that even one receptor would meet all the upper bound assumptions used, the human risk at this site attributable to cadmium exposure is deemed negligible.

Assuming a mean soil/sediment cyanide level of 100 mg/kg and an exposure scenario as just described, a total human ingestion and inhalation exposure rate of 0.214 ug/kg/visit is obtained. An acceptable daily (chronic) intake of 108 ug/kg/day has been established for human cyanide intake (USEPA, 1980); therefore, the worst case acute exposure for one visit to this site is over 500 fold lower than the level which might begin to present concerns for toxicity.

An additional avenue for human exposures is consumption of fish taken from Crab Orchard Lake. Cadmium-contaminated sediments provide a theoretical source for bioaccumulation of residues. This exposure is not quantifiable due to lack of data on fish residues and proof that residues have indeed been transported to the lake.

Wildlife Risks

Estimates of total cadmium and cyanide intakes estimated in the previous sections for receptor species of wildlife are summarized below:

Estimated Daily (Chronic) Intake - Cadmium

	Body Weight (kg)	Inhalation Rate ug/kg/day	Ingestion ug/kg/day	Total mg/kg/day
Deer	60	0.43	2.69	2.69
Rabbit	1.0	0.415	7.1	7.1
Mouse	0.03	1.0	4.75	4.75

Estimated Daily (Chronic) Intake - Cyanide

	Body Weight (kg)	Inhalation Rate ug/kg/day	Ingestion ug/kg/day	Total mg/kg/day
Deer	60	0.087	0.54	0.54
Rabbit	1.0	0.083	1.4	1.4
Mouse	0.03	0.2	0.95	0.95

NOTES: See Table 24-1 and Section 24.4.2.2 for assumptions.
Inhalation rates based on estimated breathing rates for each species, 10 mg dust inhaled per m3 of air, 0.5 ug mean Cd or 0.1 ug mean CN per mg dust, and 1 hr (burrowing animals) or 4 hrs (deer) exposure duration each contact.

These estimates indicate that wildlife exposures to site-related residues may be relatively greatest among small mammals on the landfill such as rabbits, mice, chipmunks, and the like. The latter species animals will be exposed primarily via ingestion of contaminated soil and dust while burrowing, grooming, and feeding on dust-bearing seeds and invertebrates. Rabbits and other herbivores at the site receive the major part of their exposure from contaminated vegetation. Given the broad range of demonstrated possible toxic effects, the potential for interspecific sensitivity, and limited data on effects of cadmium and cyanide on wildlife species, it is difficult to gauge the significance of these exposures. Using data from controlled tests with laboratory

animals, levels of cadmium and cyanide at this site may be sufficiently high to present concerns for reproductive effects and other systemic toxicity in vertebrate species.

U.S. EPA (1986) reports a chronic acceptable intake level for ingested cadmium to be 2.9×10^{-4} mg/kg/day, based on the lowest level which has produced kidney toxicity in humans and incorporating a ten-fold margin of safety. A small (30 g) animal consuming 5 percent of its body weight daily in the form of earthworms or comparable invertebrates might thus eat 1.5 g of food per day. If the worms contained cadmium at 100 mg/kg from living in soils with 2 mg/kg, a daily cadmium ingestion rate of 5.0 mg/kg/day can be calculated. This exposure rate is well in excess of a possible effect level. Since some sediments in the Old Refuge Shop study area contain cadmium of up to 780 mg/kg, exposure to exposed sediments combined with food chain cadmium accumulation could impact localized wildlife populations. Areas of high sediment cadmium are comparatively small, which would serve to lessen the probability for lifetime chronic exposures and overall impact of population exposures, however.

29.4.3 Analysis of Uncertainties

A principal area of uncertainty exists in addressing the risks posed to wildlife by chronic exposure to contaminants at the site. A lack of documentation on the effects of site contaminants on wildlife species which might be found on the site necessitated the use of studies involving laboratory rodents and rabbits as surrogates. The relative sensitivity of these species is unknown.

Regarding the estimate of increased cancer risk in humans from inhaled cadmium, it should be noted that the chronic lifetime exposures assumed are highly unlikely and any exposed population, chronic or otherwise, is very small. In this light, the worst case increase in human risk is considered negligible.

29.5 Preliminary Remedial Alternatives

The environmental effects discussion in Section 29.4 support the need for remedial action for the Old Refuge Shop. The contaminants identified as probable concerns were cadmium, chromium, and cyanide. Lead was detected in soil but at levels within the range for Refuge background. The contaminants were present mainly in the ditch sediment samples, although cadmium and cyanide were detected in the ground water samples and were found at concentrations above the Illinois Public Water Supply standards and Federal drinking water standards. Contamination in sediments was detected in the surface samples (approximately to 1 ft. depth), but no core samples were collected to quantify the levels in deeper sediments.

The risk-based evaluation in Section 29.4 suggests that the risk levels to wildlife species from cadmium and cyanide residues could be greater than the risks to humans, although there are limited data on the toxicological effects to the variety of wildlife species that could be exposed at the Refuge. The worst case assumptions used for the assessment of human risks determined that cadmium levels in soils were at least double the levels which would constitute acceptable risk. Cyanide levels were a concern for protection of wildlife but would not pose a risk to humans. Reducing exposure to cadmium to one half the present mean concentration, or to 250 mg/kg, would, based on the assumptions used in the risk evaluation and available data, reduce risk to

humans to a negligible level of 10^{-5}. Much lower exposure levels may be needed in order to protect wildlife, since cadmium is highly toxic to carnivorous wildlife through ingestion of contaminated dusts and invertebrate species, as well as potentially to fish if contaminants are dispersed off-site to Crab Orchard Lake. Cyanide residues are associated with sediments which contain cadmium, therefore, both contaminants will be addressed by the measures adopted to remediate cadmium. Remedial measures to address the complete transport pathways identified in the risk assessment will be given priority in the evaluation of alternatives for remedial response. Surface sediment dredging, capping, regrading, surface water diversion, and revegetating are among the technologies which will be applicable as part of this response.

The remedial objectives developed as part of the FS will be concerned with reducing risks to both humans and wildlife to acceptable levels and which will provide adequate margins of safety for protection of endangered wildlife.

Potentially applicable remedial measures for this site are discussed below.

Limited Site Access

It may be appropriate to take precautions to limit the potential for exposure to humans and wildlife via the water or direct contact with soils/sediments. Limiting site access could help prevent trespassing or unauthorized use of the site until the contaminated materials have been removed or contained. Fencing and closing the area to all but authorized personnel, and maintaining a thick vegetative cover may be appropriate until further remedial action can be initiated.

Surface Water Control

Drainage ditches or trenches might be required to divert surface water flow from areas containing elevated levels of site contaminants, in order to limit the off-site transport of contaminants.

Removal or Containment of Sediment

Contaminated surface sediments might be excavated and removed for treatment off-site or contained on-site. The contamination along the ditch extends approximately 5000 ft. towards the Lake, possibly 4 ft. wide x 2 ft. deep along the drainage route. The equivalent volume of contaminated sediment for excavation is estimated at 1600 CY. Clean fill will be used for regrading/fill for areas which may require excavation during remediation. Clean fill will be used for regrading/refilling excavated areas. Alternatively, the ditch could be drained and the sediments capped or sealed to prevent leaching of residual contaminants.

Monitoring - Surface and Groundwater

As part of the follow up remedial program, the response alternatives may include periodic sampling and analyses of the monitoring well and of the ditch water for cadmium, chromium and cyanide. Follow-up studies to verify the absence of contaminants might begin shortly after the cleanup.

29.6 Conclusions and Recommendations

It can be concluded that the Old Refuge Shop site is impacted with the primary pollutants being cadmium, cyanide, and chromium in the site's ditch water and sediment. It is recommended that remedial measures for this site should be evaluated in the FS. Potentially applicable remedial measures include limited site access to humans and wildlife and possible excavation of surface sediments.

SECTION 30 - SITE 24, PEPSI-WEST DRAINAGE

30.1 Site Description

Site 24 is a drainage ditch located north and west of the Pepsi-Cola Bottling Company building. The ditch runs parallel to the adjacent street. (See Figure 30-1). The ditch receives surface run-off from the site and discharges to Crab Orchard Lake. This site is not located on the Refuge and is not under the jurisdiction of U.S. FWS.

30.2 Site Investigations

30.2.1 Phase I Site Investigations:

One grab sample of surface water and one grab sample of sediment (0-1 ft depth) were collected from the drainage ditch.

30.2.2 Phase II Site Investigations:

One sediment sample was collected for mercury reanalysis.

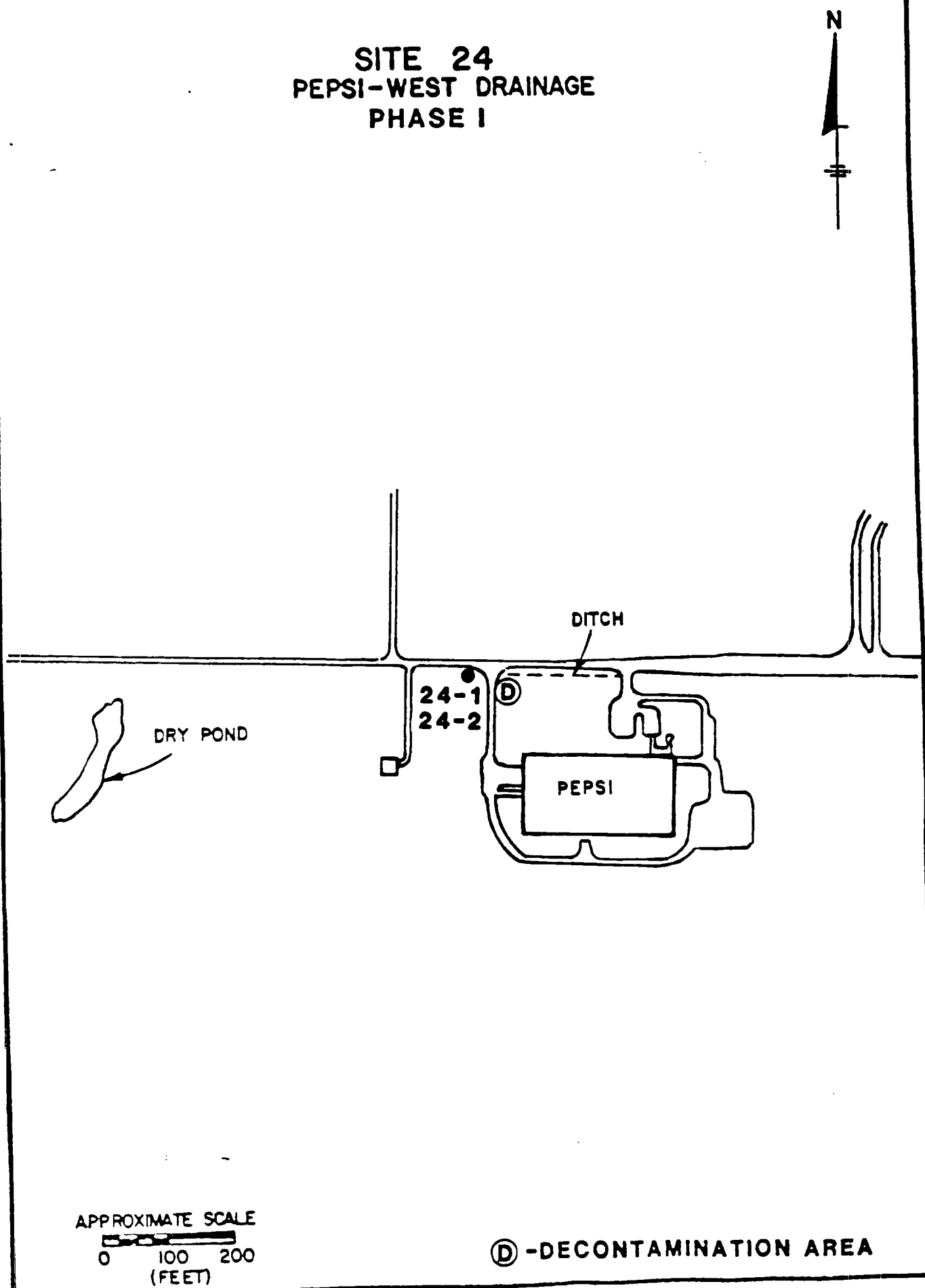
30.3 Analytical Results (See Appendix I, Page 23)

30.3.1 Phase I Analytical Results:

Total organic halide concentrations of 160 ug/L (190 ug/L duplicate) were detected in the water sample. The sediment contained acetone (268 ug/kg wet weight), and methylene chloride (117 ug/kg wet weight); however, these compounds were also detected in the laboratory blank. No other organics were detected in the sediment, although the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). The positive detections reported are thus estimated values and some compounds which

FIGURE 30-1

SITE 24
PEPSI-WEST DRAINAGE
PHASE I



O'BRIEN & GERE

were not detected may in fact be present. A mercury concentration of 9.4 ug/kg was detected, but this parameter was reanalyzed in Phase II due to questionable QA/QC support data.

30.3.2 Phase II Analytical Results:

The sediment sample contained 58 ug/kg of mercury. This concentration is greater than the background range measured at the control sites.

30.4 Environmental Effects

30.4.1 Qualitative Assessment

Sediments within the drainage ditch from Pepsi-West have been found to contain mercury at a level of 58 ug/kg. No other constituents detected in either water or sediments from this site were significantly above the levels found at the control sites. Available data do not indicate that surface water runoff from the ditch has transported mercury residues downstream to Crab Orchard Lake. Although the mercury concentration is somewhat higher than the level found at the control site soils, it is not unusual for mercury levels in U.S. soils (Lindsay, 1979). The detection of mercury at this site is not considered to be indicative of an off-site source of contamination. Since there is no waste source, a complete exposure scenario is not possible, thus it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

30.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified, there is not basis for preparing a quantitative risk evaluation.

30.4.3 Analysis of Uncertainties

The information relied upon for evaluating this location consisted of a site inspection and sample analyses. The chemical characterization data consisted of one surface water and two surface sediment samples from the ditch; deeper sediment cores or downstream water samples were not collected. Some characterization data are not sufficiently supported by QA/QC, and thus additional uncertainty is introduced dependent on the quality of the data. However, since there is no evidence to suggest that the site has been used presently or previously for disposal of wastes, and any potentially contaminated runoff from upstream areas would be transported via the surface water, the database should adequately characterize the site.

30.5 Preliminary Remedial Alternatives

The analytical results and the qualitative risk evaluation discussed in the previous sections indicate that this site does not contain contaminant levels that would result in a negative environmental impact. There will be no further evaluation of remedial alternatives, and this site will not be considered in the FS.

30.6 Conclusions and Recommendations

It can be concluded that the Pepsi-West drainage ditch site does not represent a risk of exposure to humans or to wildlife receptors. No further evaluation is recommended.

SECTION 31 - SITE 25, CRAB ORCHARD CREEK AT MARION LANDFILL

31.1 Site Description

The old Marion Landfill is located adjacent to Crab Orchard Creek on Old Creal Springs Road. This municipal landfill has been inactive for a number of years. A 3/4-acre pond is located next to the landfill. Site 25 consists of the Crab Orchard Creek sections upstream and downstream of the landfill and of the pond. (See Figure 31-1). This site is not located on the Refuge and is not under the jurisdiction of the U.S. FWS.

31.2 Site Investigations

31.2.1 Phase I Site Investigations:

One composite surface water sample and one composite sediment sample (0-1 ft depth) were collected at each of two locations approximately 250 ft. upstream and downstream of the landfill. An additional downstream sediment was also collected for CLP organics analysis. A composite surface water sample and a composite sediment sample (0-1 ft depth) were also collected from the pond.

31.2.2 Phase II Site Investigations:

One of the creek sediment samples was resampled for cyanide analysis.

31.3 Analytical Results (See Appendix I, Page 24)

31.3.1 Phase I Analytical Results:

Magnesium in the water increased from 14.3 mg/L in the upstream water sample to 47 mg/L in the downstream sample. Manganese in the

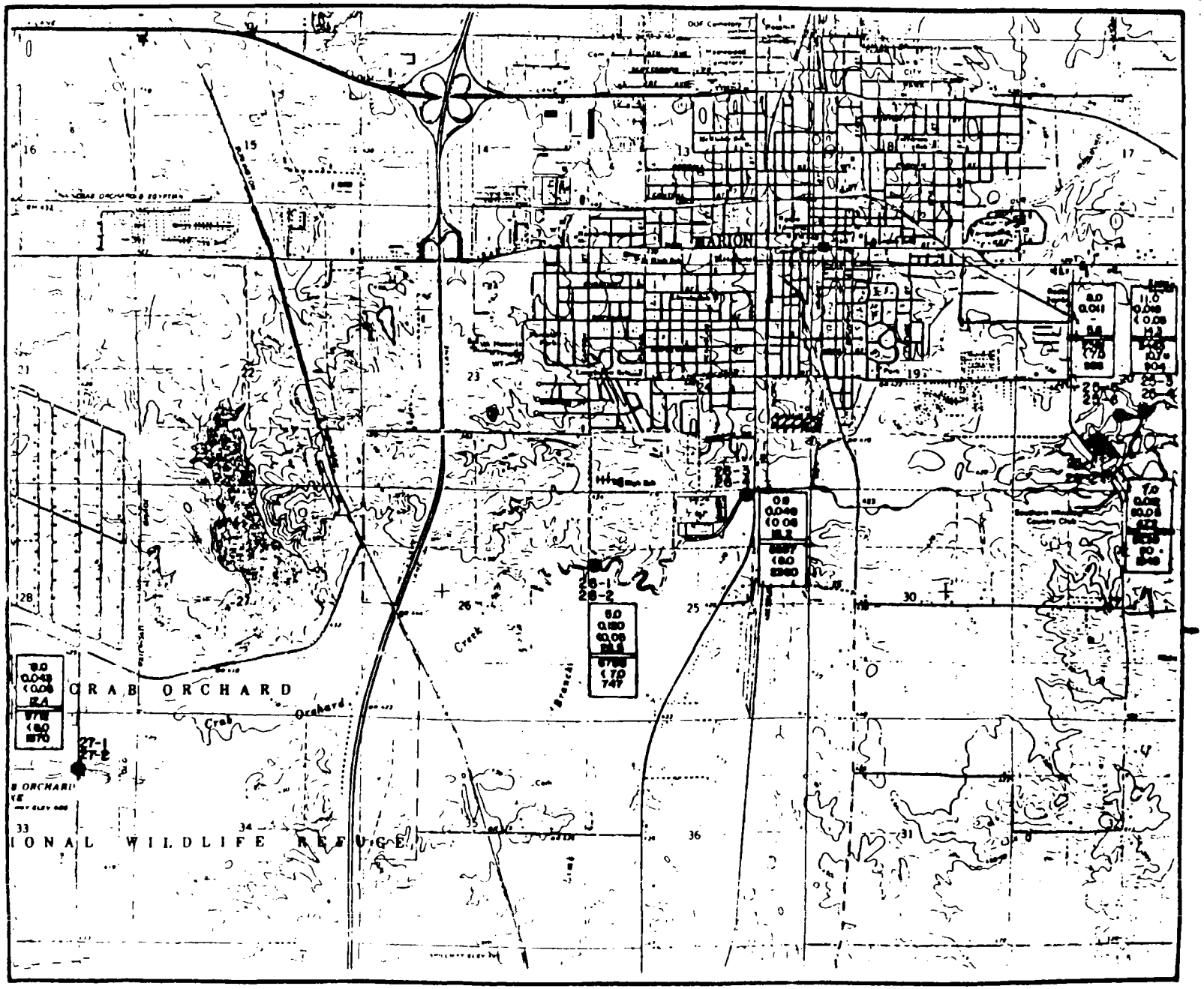


FIGURE 31-1

N

SITES 25, 26, & 27
CRAB ORCHARD CREEK
PHASE I

LEGEND

CONCENTRATIONS

WATER, mg/L

SEDIMENT, mg/kg

- PHASE II REANALYSIS

- MARION SEWAGE TREATMENT PLANT

- APPROXIMATE LOCATION OF OLD MARION LANDFILL

SCALE IN FEET

0 2000 4000

upstream water (0.68 mg/L), in the downstream water (1.5 mg/L) and in the pond (0.72 mg/L) exceeded the Federal MCL and State standard of 0.50 mg/L, but all other contaminant levels in waters were below the Illinois Public Water Supply Standards. The upstream water and pond water also exceeded the Federal MCL for iron but were at or below the Illinois Public Water Supply standard. The iron and manganese concentrations in water are not considered to pose health risks to human populations or wildlife under these conditions (see Exhibit A), since their standards are based on concerns of taste and color.

In general, the concentrations of most constituents increased in the downstream samples. Figure 31-1 shows concentrations for total organic carbon (TOC), total organic halides (TOX), cyanide and magnesium along the creek. It is noted that cyanide values are questionable due to poor QA/QC data and metals concentrations are reported as estimated values only. The creek sediments at Site 25 contained similar levels of magnesium (904 mg/kg upstream and 1,840 mg/kg downstream) as detected at the control sites. The downstream sediment contained cyanide (90.4 mg/kg) compared to below 8 mg/kg upstream, and elevated TOC (18,239 mg/kg) compared to 3,778 mg/kg upstream. The pond sediment contained magnesium (956 mg/kg), and below 7 mg/kg cyanide. No priority pollutants were detected in the CLP analysis, although FID scans in the sediments ranged from 3,735 ug/kg to 220,368 ug/kg.

31.3.2 Phase II Analytical Results:

The cyanide concentration in the creek sediment upstream of Site 25 was 10.7 mg/kg.

31.4 Environmental Effects

Environmental Effects for sites along Crab Orchard Creek are discussed in Section 33.4.

31.5 Preliminary Remedial Alternatives

Preliminary Remedial Alternatives for sites along Crab Orchard Creek are discussed in Section 33.5.

31.6 Conclusions and Recommendations

Conclusions and Recommendations for sites along Crab Orchard Creek are discussed in Section 33.6.

SECTION 32 - SITE 26, CRAB ORCHARD CREEK BELOW MARION STP

32.1 Site Description

The Marion Sewage Treatment Plant (STP) discharges to Crab Orchard Creek upstream of Court Street in the Village of Marion. This site is not within the boundaries of the Refuge and is not under the jurisdiction of the U.S. FWS.

32.2 Site Investigations

32.2.1 Phase I Site Investigations:

One composite surface water sample and one composite sediment sample (0-1 ft depth) were collected at each of two locations, both downstream of the treatment plant, to assess the impact on various segments of the creek. The first sampling location was at the intersection of the creek and So. Carbon Street (samples 26-1, 26-2). The second sampling location was at the intersection of the Creek and Court Street (samples 26-3, 26-4). The sampling locations were spaced approximately 2,000 ft. apart (see Figure 31-1).

32.2.2 Phase II Site Investigations:

No samples were collected in Phase II.

32.3 Analytical Results (See Appendix I, Page 25)

The FID scan, lead, magnesium, and zinc concentrations were approximately one order of magnitude higher in the downstream sediment samples than in the upstream samples, but all concentrations were similar to those detected at the control sites. The metals concentrations are included as

estimated values for screening purposes only (see Exhibit B). Chloroform was detected in both the upstream and downstream water samples at 2 ug/L, exceeding the AWQC for human health of 0.19 ug/L, but is well below the same criterion set for aquatic life protection of 1,240 ug/L. No other volatile organics or PCBs/Pesticides were detected in either the water or sediments although TOX and FID scans detected organics. Semivolatiles were not analysed at this site.

TOX levels were somewhat elevated in the downstream water sample, 120 ug/L (130 ug/L duplicate) versus 49 ug/L upstream. Manganese concentrations in the upstream and downstream waters (0.3 and 0.75 mg/L) both exceeded the Federal MCL and Illinois Public Water Supply standards. Iron in the downstream sample (1.0 mg/L) also exceeded the Federal MCL, but not the Illinois standard. Iron levels are regulated due to objectionable taste and color; at these levels, this does not pose a threat to public health. All other surface water parameters were within Illinois Public Water Supply Standards. Figure 31-1 shows the concentration for TOC, TOX, cyanide and magnesium in waters and sediments from Crab Orchard Creek. All parameters in the upstream location (26-1, 26-2) were below the levels found upstream of the STP (Site 25), with the exception of TOX.

32.4 Environmental Effects

Environmental Effects for sites along Crab Orchard Creek are discussed in Section 33.4.

32.5 Preliminary Remedial Alternatives

Preliminary Remedial Alternatives for sites along Crab Orchard Creek are discussed in Section 33.5.

32.6 Conclusions and Recommendations

Conclusions and recommendations for sites along Crab Orchard Creek are discussed in Section 33.6.

SECTION 33 - SITE 27, CRAB ORCHARD CREEK BELOW I-57 DREDGE AREA

33.1 Site Description

Site 27 is located in Crab Orchard Creek downstream of the Interstate Route 57, approximately 4,000 ft. downstream from Site 26 (See Figure 31-1). Dredging of the stream bed was conducted in this area a number of years ago. This site is located on the Refuge.

33.2 Site Investigations

33.2.1 Phase I Site Investigations:

A composite surface water sample and a composite sediment sample (0-1 ft depth) were collected from Crab Orchard Creek where the creek intersects with Chammness Road.

33.2.2 Phase II Site Investigations:

No samples were collected in Phase II.

33.3 Analytical Results (See Appendix I, page 26)

TOX levels in the surface water of 43 ug/L (38 ug/L duplicate) were below those found at Site 26. Most other parameters were similarly below the levels detected at Site 26 except for TOC and magnesium in the sediment. Figure 31-1 shows the concentration for TOC, TOX, cyanide and magnesium along Crab Orchard Creek. The cyanide results were questioned due to poor QA/QC and the metals concentrations are estimated values (see Exhibit B). The manganese concentration (0.64 mg/L) in the water exceeded the Federal MCL and Illinois Public Water Supply Standard as it did in each of the five samples collected further upstream. Iron (0.5 mg/L) also exceeded the Federal

MCL but not the Illinois standard. The concentrations of all other contaminants in the water were below the Illinois Public Water Supply Standards and Federal MCLs. The excursions noted for iron and manganese do not represent a health concern since these standards are implemented based on aesthetic considerations of taste and color.

Sediment concentrations were similar to the ranges detected for the Refuge control sites.

33.4 Environmental Effects

33.4.1 Qualitative Assessment

33.4.1.1 Source Evaluation

This section is an analysis of exposure and risk in Crab Orchard Creek at the Marion Landfill (Section 31), below the Marion Sewage Treatment Plant (Section 32), and below the I-57 Dredge Area (Section 33).

33.4.1 Qualitative Assessment

33.4.1.1 Source Evaluation

Crab Orchard Creek At Marion Landfill

Crab Orchard Creek may receive leachate and runoff from the Old Marion Landfill, now inactive. Analyses of the site revealed magnesium residues up to 47 mg/L in the creek water and up to 1,840 mg/kg in the sediment of a nearby pond; iron concentrations in water up to 1,000 ug/L, manganese levels in water up to 1,510 ug/L, and cyanide in creekbed sediment of 10.7 to possibly 90.4 mg/kg were also found. Exceedance of the standards for manganese and iron do not represent a health concern since these criteria were established based on aesthetic considerations.

The reported maximum cyanide concentration of 90 mg/kg is estimated due to deficiencies in QA/QC support data from the Phase I analysis. Since no other water or sediment samples along Crab Orchard Creek contained elevated levels of cyanide, this value seems to represent a false positive, and cyanide will not be considered as a site contaminant. Magnesium levels are within the ranges typically found in soils (Lindsay, 1979). Due to the lack of source of exposure, a complete scenario is not possible and this site will not be considered further.

Crab Orchard Creek Below Marion Sewage Treatment Plant

No water or sediment cyanide levels were detected at this site that would constitute a threat to human health or the environment. Iron and manganese levels in water were above the aesthetic-based standards but are not considered to represent a health concern. Due to the lack of source of exposure, this site will not be considered further.

Crab Orchard Creek Below I-57 Dredge Area

As discussed in Section 33.3, water sampled from the creek contained iron and manganese levels above the Illinois Public Water Supply and Federal drinking water standards. Iron and manganese levels were above the MCL set for aesthetic concerns but were within the standards for health protection. Sediment levels were comparable to the Refuge control sites. Magnesium in the creek water ranged from 5.6 to 47.2 mg/L. Since there is an insufficient data base from which to determine risks to fish, wildlife, and humans at this site, and since federal criteria or standards for magnesium have not been promulgated, a receptor analysis and quantitative risk assessment will not be performed.

33.4.3 Analysis of Uncertainties

Several areas of uncertainty can be noted in the evaluation of Sites 25, 26, and 27 along Crab Orchard Creek:

1. The limited number of samples collected may not completely characterize the sites, given the large geographical area studied;
2. The uncertainties inherent in some of the laboratory analyses due to inadequate QA/QC support data may underestimate the presence of site contaminants;
3. The chemical form and toxicity of cyanide residues, if any, have not been determined.

33.5 Preliminary Remedial Alternatives

Follow-up monitoring of surface water and sediments is suggested. Attachment 1 details a recommended monitoring program. The parameters identified as possible targets for monitoring along Crab Orchard Creek sites include total organic carbon, total organic halides, cyanide, manganese, iron, and magnesium.

33.6 Conclusions and Recommendations

The water samples from Crab Orchard Creek exceeded the State and Federal standards for iron and manganese, however these standards were established based on aesthetic concerns of taste and color, therefore, the levels found do not pose a threat to human health or wildlife. Sediment contaminant levels detected were not supported by QA/QC data and do not appear to have leached to the water. It can be concluded that follow-up monitoring studies (see Attachment 1) will adequately characterize the conditions of the site in the event that these conditions change.

SECTION 34 - SITE 28, WATER TOWER LANDFILL

34.1 Site Description

Historical aerial photographs indicate that landfilling activities occurred at Site 28, adjacent to the Water Tower near Areas 7 and 14. (See Figure 34-1).

The photographs from 1943 show that the area encompassed by the Water Tower road is radially marked by tire tracks, and debris can be seen at the end of the tracks. The landfill appears as a diamond-shaped area to the north of the Water Tower. Photographs from 1951 show that the landfill was not in use at that time, and the area previously noted to be marked with tire tracks is now vegetated and the landfill area is no longer distinguishable. The landfill appears to have been reactivated sometime after 1951, as debris and equipment are shown in aerial photographs from 1960 and 1965. By 1971, the landfill is no longer active and the area is partially vegetated. The areas of site activity indicated on the 1943 and 1965 air photos are illustrated on Figure 34-1.

These activities are not visually apparent today; however, a number of rusted drums, metal parts and tar residues are present. The site gradually slopes to the northeast. The sloping face northeast of the Water Tower is heavily overgrown with briars and rutted with several major gullies; only a small amount of refuse is evident in this area. More activity is evident in the woods at the bottom of the slope. Standing water in the main drainage gully showed a slight sheen on the surface on one site inspection, but was not evident in latter visits to the site. This gully ultimately discharges approximately 1 mile northward to Crab Orchard Lake. Several small mounds are within the woods and a larger mound is located at the top of the hill. Previous soil sampling by DOI detected lead concentrations up to 800 mg/kg. (Ruelle, February 1983).

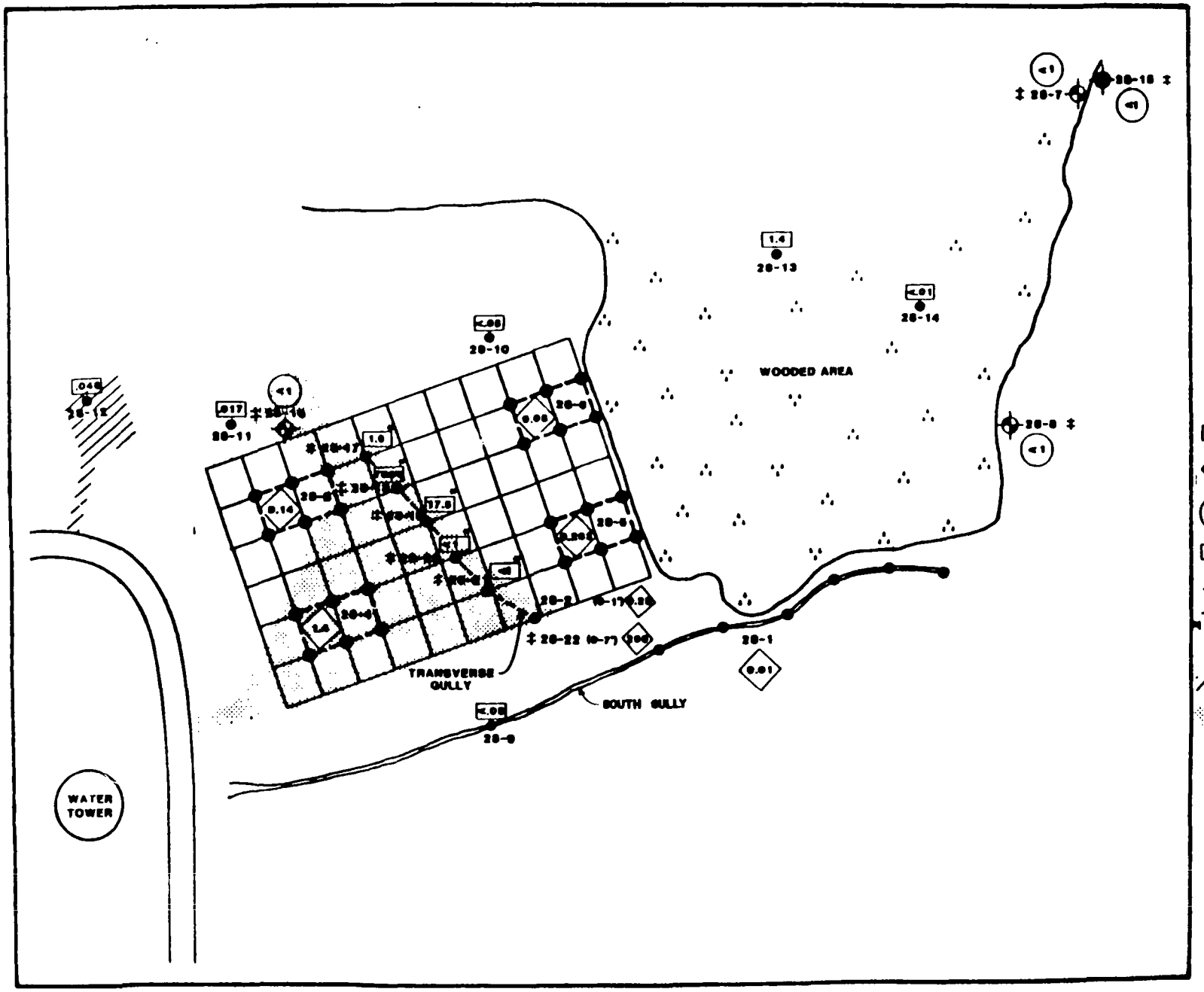


FIGURE 34-1

N

SITE 28
WATER TOWER LANDFILL
PHASE I & II

LEGEND

PCB CONCENTRATIONS

GRAB SAMPLE SOIL, mg/kg WET WEIGHT

AREAL SURFACE COMPOSITE SOIL, mg/kg WET WEIGHT

WELL, ug/L

SOIL, mg/kg WET WEIGHT 0-1' TEST PIT

SHALLOW WELL

DEEP WELL

PHASE II SAMPLES

- AREA OF SITE ACTIVITY INDICATED ON 1943 AIR PHOTO

- AREA OF SITE ACTIVITY INDICATED ON 1988 AIR PHOTO

SCALE IN FEET

0 50 100

34.2 Site Investigations

34.2.1 Phase I Site Investigations:

A magnetometer and electromagnetic terrain conductivity survey was conducted along and transverse to the slope of the landfill. (See Figures 34-2 and 34-3). Two shallow ground water monitoring wells were also installed and set at depths of 20 and 25 feet in clayey silt. Both wells included ten foot length well screens from intervals of 10-20 and 15-25 feet respectively.

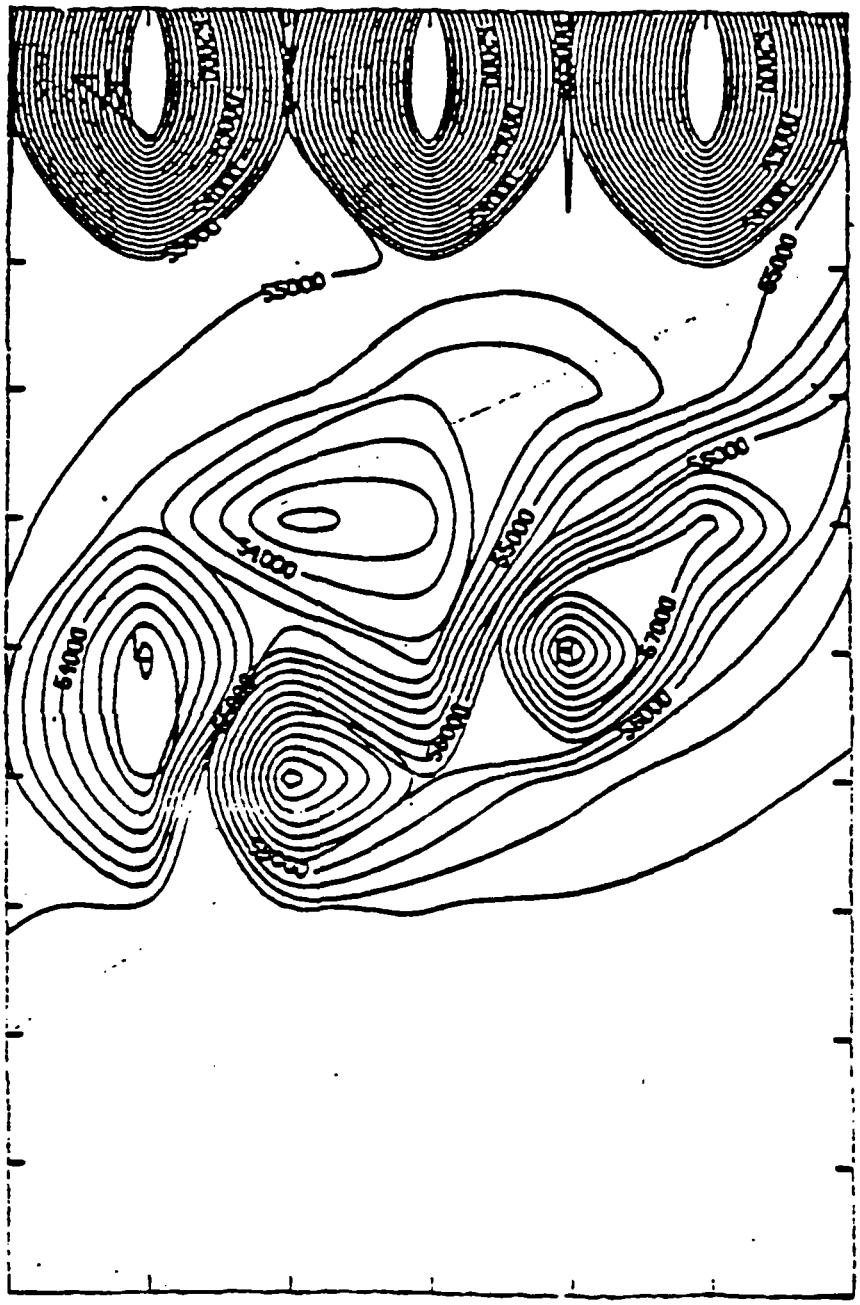
A survey grid was established for the area northeast of the Water Tower where burial activities may have occurred based on the aerial photographs from 1965. Composite soil samples each consisting of six, 0-1 ft. depth grabs) were collected along a central transverse drainage gully, the Water Tower drainage ditch (south of the grid), and from grids spaced toward the outer edges of the grid. Six additional grab soil samples (0-1 ft. depth) were collected from other locations, including two from the diamond shaped area to the north of the Tower, two within the wooded area where scattered debris was found, and two from locations removed from the grid. One sample from the transverse gully was resampled for full CLP organics analyses.

34.2.2 Phase II Site Investigations:

Phase II site investigations included the installation and sampling of a set of nested ground water monitoring wells in addition to the sampling of the two wells installed in Phase I. The Phase II wells were set at depths of 20.5 and 39 feet and screened from 15.5-20.5 and 34-39 feet respectively. Soils encountered at these sites were generally silt and silty clay with a fine sand layer identified at 34-39 feet in the deep well, Well

SITE 28

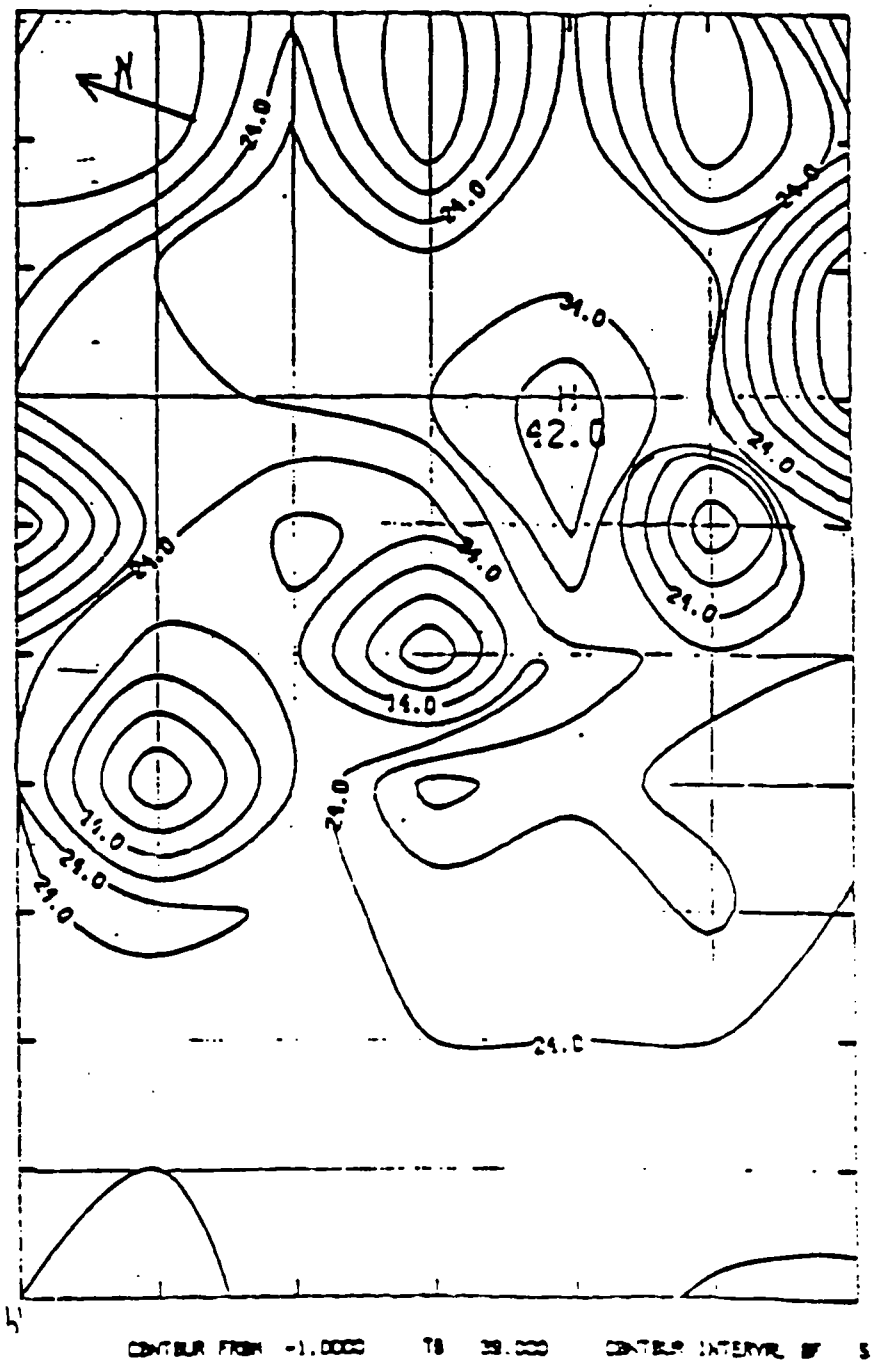
MAGNETOMETER SURVEY



CONTOUR FROM 50000. TO 58500. CONTOUR INTERVAL OF 250.00

SITE 28

ELECTROMAGNETIC SURVEY



28-15. The ground water samples were analyzed for CLP HSL volatiles, pesticides, PCBs, and metals, as well as for cyanide and indicators.

Five soil test pits were dug to a depth of 7 feet. The test pit locations were selected based on the data results from the geophysical surveys. The pits were dug to explore subsurface soils, although the Phase I screening did not show elevated contaminants in any of the surface soils. Six soil samples were collected from these pits: one composite (0-7 ft. depth) from each pit, as well as one composite combining the five test pit samples (See Figure 34-1). The soils were analyzed for PCBs, magnesium, copper, lead, arsenic, and cyanide.

34.2.3 Site Hydrogeologic Characterization

34.2.3.1 Site Geology

Based on results of the well installations at Borings 28-7, 8, 15 and 16, the subsurface unconsolidated overburden consists of a mottled grey, orange, and brown silty clay to clayey silt containing trace amounts of sand and fine gravel. This material continues vertically to depths of 20 to 25 ft. below ground level, and appears to be laterally continuous throughout the site. Beneath the silty clay layer, approximately 12 ft. of clay is present to a depth of 24 to 36 ft. as exhibited in the deep boring 28-15. A thin layer of fine-coarse sand is present beneath the clay, and occurs above a medium grained, light grey sandstone bedrock. Top of bedrock is 37.5 ft. below ground level in the deep boring. As only one boring encountered materials beneath the silty clay layer, no estimate can be made of the lateral extent of the lower unconsolidated layers.

34.2.3.2 Site Hydrogeology

Occurrence of Ground Water

Shallow ground water occurring beneath the site was found at a depth of 1.5 to 18 ft. below the ground surface within the silty clay/clayey silt soil unit. The three shallow ground water monitoring wells were screened in this upper water table. Ground water elevations collected during the winter and summer of 1987 (wet and dry seasons, respectively) indicate a water table fluctuation of 2 to 4 ft. with water levels dropping during the summer months (Table 4-3).

A lower ground water aquifer was encountered in the lower portions of the soil sequence on top of the sandstone bedrock where unconsolidated sands were encountered. The deep well installed on site was screened in this lower aquifer. Ground water elevations collected during the winter and summer of 1987 indicate a 0.75 ft. fluctuation in the water table with levels dropping in the summer months (Table 4-3). This lower aquifer is confined, as is apparent by a 3 ft. higher elevation in ground water in the deep well as opposed to that occurring in an adjacent nested shallow well.

Ground Water Flow Conditions

Ground water elevations from the shallow ground water monitoring wells were contoured and are presented in Figure 34-4. The ground water flow direction is toward the northeast, i.e. along a north-south trending stream which flows north into Crab Orchard Lake. The hydraulic gradient of flow (i) during June 18-19, 1987 was approximately 0.016 ft/ft. The average hydraulic conductivity (K) for the shallow wells was calculated to be about 0.37 ft/day. Porosity was assumed to be 0.35 (Davis and Dewiest).

A calculation was then made of the average ground water flow velocity (Vs) through the upper soil units. Using the formula given in Section 4.2, the resultant velocity was calculated to be about 0.017 ft/day or 6.2 ft/year. The flow velocity is controlled by the relatively low hydraulic gradient and low hydraulic conductivity occurring in this area. An upward vertical flow was identified from the unconsolidated aquifer screened by the deep Well 28-15 into the upper water table. This phenomenon indicates discharge of deep ground water towards Crab Orchard Lake.

34.3 Analytical Results (See Appendix I, Page 27)

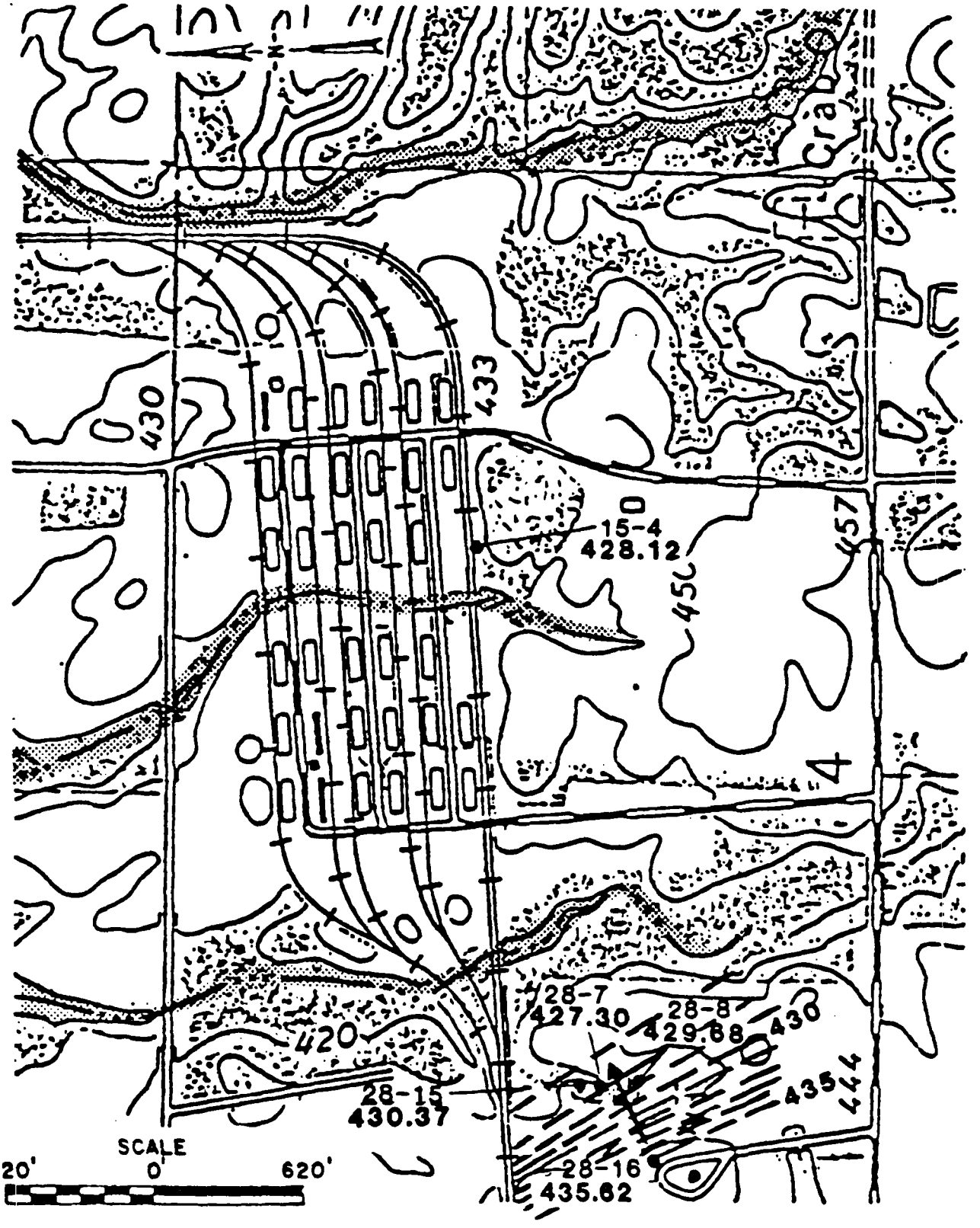
34.3.1 Phase I Analytical Results:

The magnetometer and electromagnetic surveys shown in Figures 34-2 and 34-3 suggested the presence of metallic debris within the transverse ditch. Some scattered residues of metallic debris were found during the site inspection within the wooded area to the east.

The symmetrical contour lines shown in Figure 34-2 north east of the grid resulted from the more narrow grid spacing which had to be used within the wooded area, and was not due to any metallic or conductive objects at the Site. Figure 34-1 shows the detected PCB concentrations on a wet weight basis. Two of the soil samples (28-4, a surface grid composite, and 28-13, a grab from the wooded area) contained 2.8 and 1.7 mg/kg dry weight respectively, which were higher than those detected in the other soils (less than 0.01 to 0.354 mg/kg dry weight). The soil composite collected from the transverse gully (sample 28-2) at a 0-1 ft depth contained 0.35 mg PCBs/kg. The soil composite along the ditch south of the landfill area contained 0.023 mg PCBs/kg dry weight.

FIGURE 34-4

SITE 15 & 28 GROUNDWATER FLOW MAP



SITE 15 & 28
15 - ACID POND
18 - WATER TOWER
-STREAM

28-7 JUNE 18, 1987
● - MONITORING WELL LOCATION & IDENTIFICATION
427.30 - GROUND WATER ELEVATION
430 - - - GROUND WATER CONTOUR

O'BRIEN & GERE ENGINEERS INC

Lead was detected in all of the surface soil samples at concentrations ranging from less than 20 to 250 mg/kg. Lead concentrations are estimated and are reported for screening purposes only. The highest lead concentration was detected at Location 28-4, which also contained the highest PCB concentration. The range of lead concentrations measured would not be considered atypical for lead levels in soils.

34.3.2 Phase II Analytical Results:

The soil samples collected along the transverse gully (Figure 34-1; location 28-2, 0-1 ft composite for Phase I, and locations 28-17 to 28-22 at 0-7 ft depth for Phase II), contained PCB concentrations in the range of less than 1 to 22 mg/kg for four test pits, and one high concentration of 8,900 mg/kg at test pit location 28-18. The composite of the five test pits (Sample 28-22) contained 320 mg/kg PCBs, skewed by the detection of residues in one test pit. Lead concentrations ranged from 13 to 712 mg/kg for the test pit soils with the exception of sample 28-18 which contained 4,300 mg lead/kg. Copper concentration was also elevated (8.4 to 813 mg/kg) compared to the Refuge background. The highest copper value was detected in the sample from 28-19, adjacent to the pit with elevated PCBs and lead. Magnesium levels were found at 2,940-16,300 mg/kg which are within an order of magnitude of the Refuge background.

The ground water samples all contained below 1 ug/L PCBs. Traces of chloroform were detected in nested wells 28-15 and 28-16 only, at 2 and 10 ug/L, respectively. Acetone and methylene chloride were reported in the waters but their presence is attributed to laboratory contamination. No pesticides were detected.

All dissolved metals concentrations were below the Illinois Public Water Supply Standards and Federal MCLs and MCLGs; however, only the unfiltered samples were analyzed for wells 28-15 and 28-16. The concentrations of some metals were detected above the standards for these unfiltered well samples. Total copper was 117 ug/L in Well 28-16, compared to the Federal Drinking Water standard of 1,000 ug/L, and the Illinois State standard of 20 ug/L. Manganese exceeded the Illinois and Federal MCL standard of 50 ug/L for the same well at a concentration of 2,780 ug/L total. Total iron in the monitoring wells exceeded the Federal MCL standard for iron of 300 mg/L, with concentrations between 425 ug/L and 94,600 ug/L. It should be noted that the excursions for iron and manganese are not considered to represent a concern for public health or wildlife protection, since these standards are promulgated due to considerations of taste and color. Unfiltered lead (4.5-76 ug/L) and zinc (16-355 ug/L) were also elevated, but only lead was above the Federal standard of 50 ug/L.

34.4 Environmental Effects

34.4.1 Qualitative Assessment

34.4.1.1 Source Evaluation

Although aerial photographs indicate that a waste disposal area existed in the vicinity of the Water Tower, there is no information currently available regarding the history or source of the wastes disposed of at this site. Most information regarding the nature and composition of wastes in this area is based on the test pit investigation. The test pit investigation, which was based on the results from the geophysical surveys, revealed that the metallic objects in the landfill were not

directly identifiable as being of municipal origin, but appeared to be small pieces of stamped metal, wire and other objects. It is presumed that these materials may have originated from one or more of the industrial sites in the area.

There were no containers or containerized wastes located in any of the test pits, nor were any such materials located on the surface of the site, with the exception of one empty rusted drum in the wooded area. Likewise, there were no liquid or semi-liquid wastes, such as oils or tars, observed in the test pits or surfaces of the site. The primary waste materials encountered were what appeared to be wires, small metal parts and small electrical components, such as switches. There were no solid chemical wastes observed in the test pits or at the surface of the site. At this time, with the exception of the test pit areas, the site is covered by vegetation.

Based on the results of the surface soil sampling, the surface of the site appears to be free from waste contamination. The results of the test pit investigation indicate the presence of PCBs and lead at one of the subsurface locations. With the exception of PCBs, priority pollutant organics were not significantly different from those detected at the control sites.

Although all of the components of waste at this site are not known, based on the chemical compounds detected (PCBs, lead and copper), the waste residues are not especially mobile, and would therefore be expected to remain in place within the landfilled area. Due to low vapor pressures characteristic of these compounds, these compounds do not tend to volatilize. Also, PCBs have a very low solubility in water and a high organic carbon adsorption coefficient. These two factors significantly retard the leaching of PCBs into the ground water.

Lead and copper also have a limited potential for mobility. Depending on the type of compound they are a component of, these two metals demonstrate only low to moderate solubilities in water. Also, as cations, they become adsorbed to active (predominantly negative) surfaces on soils and sediments, thereby retarding their ability to be leached and be transported with ground water.

Of the materials detected within the site, PCBs have the highest potential to cause toxic effects. PCBs have been demonstrated to cause both acute and chronic toxic effects at low doses. Acute effects are of most concern to fish and wildlife (Exhibit A). PCBs have also been demonstrated to be carcinogenic in animal studies. In comparison with the toxic properties of PCBs, lead and copper are considered as minor toxic components at the levels at which these metals were detected at the site. The PCBs will therefore be the major source compound considered in this risk assessment.

34.4.1.2 Transport Route Evaluation

a) Air: Because the waste materials are currently covered by a layer of vegetation and have not been shown to be present in samples of surface soils, the waste materials present at this location are not considered to be a significant source for release to and transport by the air route. The site contaminants (PCBs, lead and copper) do not exert an appreciable vapor pressure and will not diffuse out of soils into air at significant concentrations. For these reasons, the air route is concluded to be non-functional at this location.

b) Direct Contact: As with the air route, the direct contact route cannot be considered to be complete, based on the absence of

exposed waste materials. However, in the event that portions of the site cover are disrupted, such as by excavation or by burrowing activities by small mammals, wastes may become exposed, thus creating a source for the direct contact pathway. However, in its present condition it can be assumed that the direct contact route is not functional at this location.

c) Surface Water: Based on the absence of exposed waste materials that could come in contact with surface water runoff at or near the site, it can be concluded that the surface water transport route is not functional at this location. However, should the waste materials become exposed, they could be scoured by precipitation and runoff, thereby entering the surface water.

d) Ground Water: The results of the groundwater sampling and analysis program indicate that waste components are not present in groundwater above the limits of detection. The measured soil hydraulic conductivity at this site ranges from 2.48×10^{-7} to 5.93×10^{-6} ft/sec (Table 4-4). The transport of PCBs and lead are significantly retarded by this type of soil. On this basis, and the results of the ground water analyses which indicate the absence of waste components, it can be concluded that the ground water transport route is not functional.

34.4.1.3 Receptor Evaluation

Human

Based on the results of the site inspection and analytical program, and the fact that the waste materials are currently covered by soils and vegetation, the transport route evaluation indicated that there is

currently no potential route by which human receptors in the area may become exposed to the waste materials. However, should this area be excavated in the future for construction or other purposes, all of the routes, with the exception of the groundwater route, would become functional and the potential for human exposures would arise.

Wildlife

The Water Tower Landfill is in a wildlife refuge and therefore the potential for contaminant exposure is also possible for wildlife. As with human exposures, the fact that the waste materials are currently covered with vegetated topsoil generally provides a barrier against exposures by wildlife. However, exposures may be experienced by burrowing animals if these were to establish dens within the area of the fill. Nevertheless these exposures would be minimal, based on the results of the test pit investigation, which identified PCBs and metals in isolated areas, rather than evenly distributed throughout the site.

34.4.2 Quantitative Assessment

Because no complete exposure scenarios could be identified in the qualitative risk assessment, there is no basis for preparing a quantitative risk evaluation.

34.4.3 Analysis of Uncertainties

The sampling locations for this site were based on reviews of historical aerial photographs, geophysical surveys, and site inspections. This information was used both for selection of surface and deep soil sampling locations. Based on the results of these surveys, in addition to

the Phase I and Phase II analytical sampling programs for soils and ground water at this site, it is concluded that the data generated should be adequate to characterize the site.

34.5 Preliminary Remedial Alternatives

The Phase I and Phase II sampling program for this site did not detect surface soil contamination or contaminant migration via ground waters. All constituents analyzed in the surface samples were within the ranges found at the Refuge control sites. Deeper soil samples (1 to 7 ft. depths test pits) from one area revealed a localized source of PCBs and lead. However, based on the overall evaluation of site conditions, hydrogeology, ground water analyses, surface and subsurface soil characteristics, the conclusions of the risk evaluation are that the waste source is adequately isolated, and no complete pathways were identified that might pose a risk to wildlife or humans under present site conditions.

Based on the conclusions derived from the risk evaluation, potentially applicable remedial efforts to be evaluated in the FS might include ground water monitoring, limited site access, capping, surface water diversion, regrading, and revegetating. Site characterization data generally support these remedial technologies; however, in the event that excavation and transport of materials is considered, additional sampling might be conducted to more accurately define the waste source. Estimates for evaluation of remedial response alternatives in the FS will be made based on available data including aerial photographs, geophysical surveys, and the RI sampling results.

A summary of potential remedial actions for this and all sites at the Refuge is shown in Table 2 of the Executive Summary. The remedial technologies will be reviewed in more detail as part of the FS.

Any considerations for future land uses should be evaluated and approved by the Refuge Management. Some of the potentially applicable remedial measures for this site are reviewed below.

Limited Site Access

It may be advisable to reduce human and/or wildlife access to the Water Tower site. Construction of a fence around the landfill area would accomplish this, although maintaining a thick vegetative cover may provide adequate protection since wastes are currently not exposed. Deed restrictions may be incorporated to control future uses of the site.

Capping, Regrading, Control of Surface Water

To further limit the potential for exposure to subsurface contamination, engineering controls such as capping may be instituted at this site. Regrading and diversion of surface streams and drainage channels may be implemented in lieu or in conjunction with the cap, to protect surface soil erosion.

Monitoring - Groundwater

The remedial response actions may include periodic sampling and analyses of the four monitoring wells for PCBs and lead.

34.6 Conclusions and Recommendations

It can be concluded that the Water Tower Landfill site contains subsurface soil contaminants with the primary pollutants being PCBs and lead. It is recommended that remedial alternatives for this site be evaluated in the FS. Since the sampling results confirmed only subsurface contamination, remedial efforts will focus on preventing future exposure to human or wildlife receptors. Potentially applicable remedial measures to be evaluated in the FS include regrading, surface water diversion, drainage controls, capping, limited site access, and ground water monitoring.

SECTION 35 - SITE 29, FIRE STATION LANDFILL

35.1 Site Description

Site 29 is a large open field (roughly 350 ft. x 300 ft) southwest of the Refuge Fire Station. (See Figure 35-1). The field was used for storage of mining machinery until several years ago. Prior to that, the landfill was reportedly used by Olin, and a fire is reported to have occurred. Debris is visible on the northern and eastern embankments. The eastern edge of the landfill drops four or five feet to a swampy area. Much of the debris consists of concrete, metal, wire, and other machinery-related items. A slight sheen was observable on the swamp water during one site inspection. Previous soil sampling by DOI on the north side of the field detected lead concentrations up to 553 mg/kg (Ruelle, February 1983). The landfill is suspected to contain ignitable magnesium metal, according to the Refuge Manager. An empty 30-gallon drum labelled "Magnesium Powder" was found along the southern end of the eastern face.

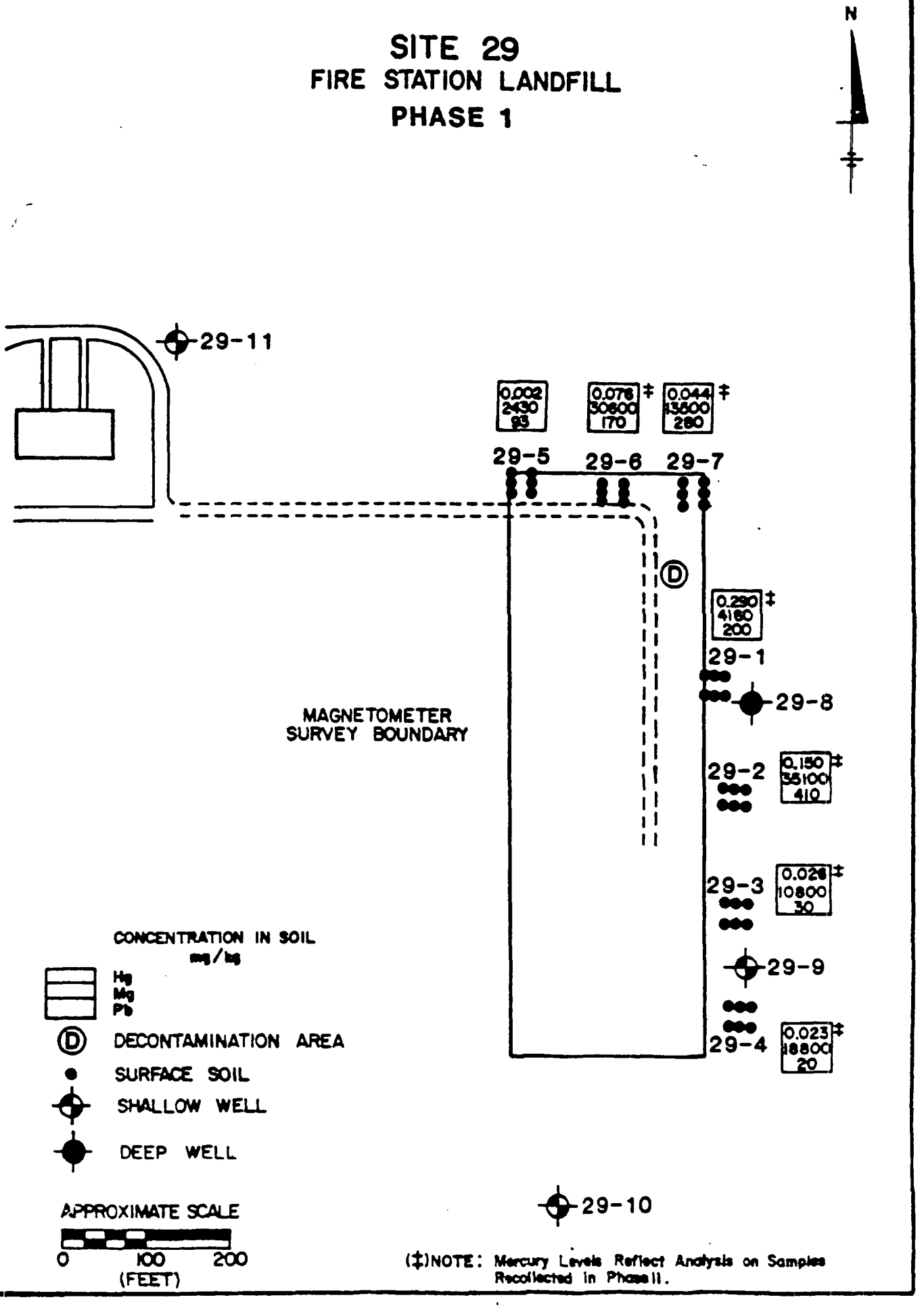
35.2 Site Investigations

35.2.1 Phase I Site Investigations:

A magnetometer and electromagnetic terrain conductivity survey was conducted over the northeastern end of the field using grid spacings of 20 feet. (See Figures 35-2 and 35-3).

Four ground water wells were installed. Three shallow wells were set within silty clay and silty sands to depths of 15, 25 and 30 feet. The fourth well was installed at a depth of 23 feet in sandstone bedrock. All wells screened the lower 10 feet of the boring.

SITE 29
FIRE STATION LANDFILL
PHASE 1

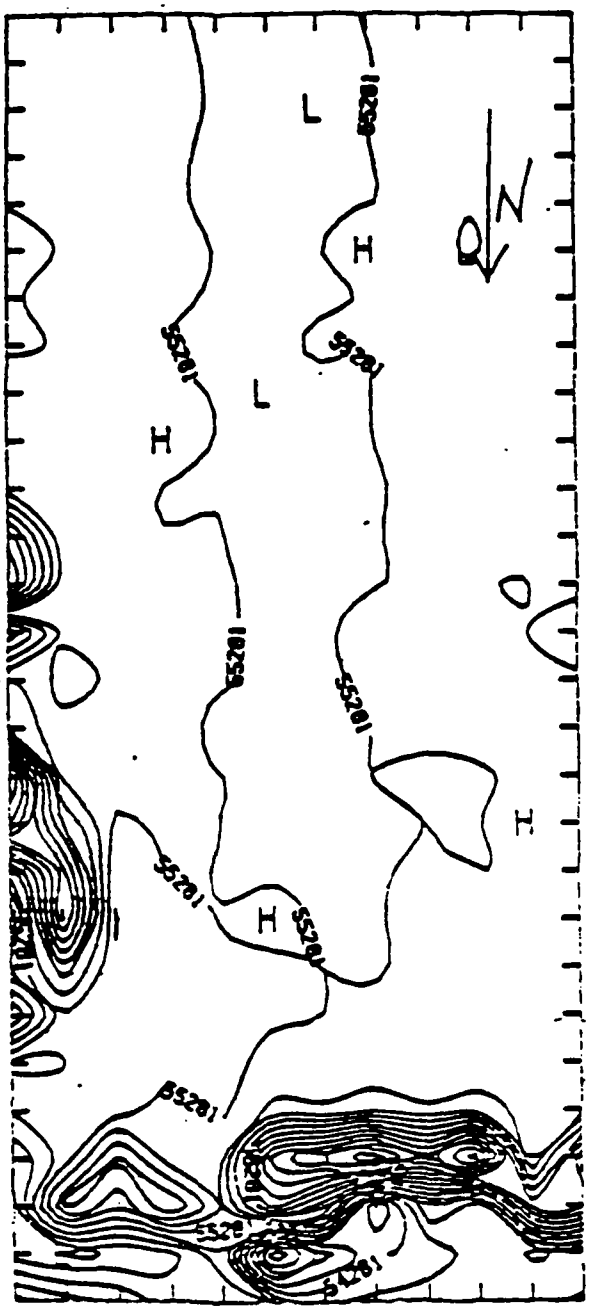


(‡)NOTE: Mercury Levels Reflect Analysis on Samples Recollected in Phase II.

O'BRIEN & GERE ENGINEERS INC

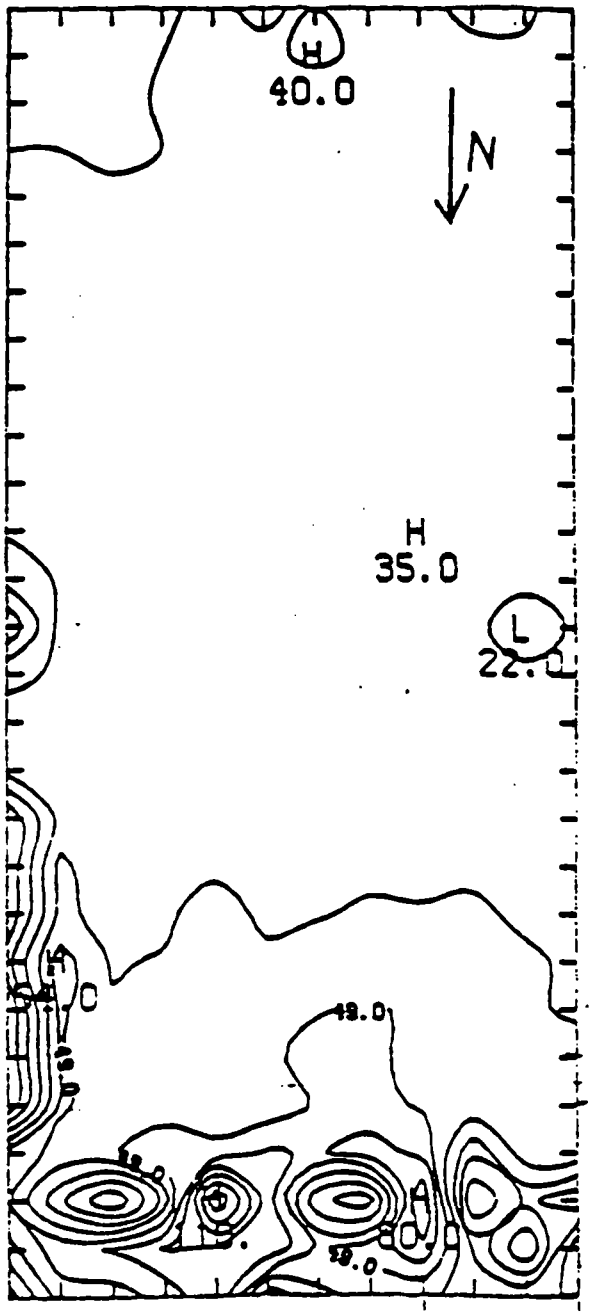
SITE 29

MAGNETOMETER SURVEY



SITE 29

ELECTROMAGNETIC SURVEY



O'BRIEN & GERE ENGINEERS INC

Seven composite soil samples (12 grabs each, at 0-1 ft depth) were collected at grid locations along the eastern and northern faces of the field. One soil from the eastern face was resampled for full organics analysis.

35.2.2 Phase II Site Investigations:

Ten test pits were dug to a depth of six feet, five pits along the east face and five along the north face. One composite soil sample was collected for each of the test pits along the east and north faces. Shallow (0-1 ft. depth) soil samples 29-1 to 29-7 were resampled at the Phase I locations for mercury analyses. Soil samples were also collected from the ditch approximately 50 feet from the landfill edge to prepare a field composite for the north and east ditch transects. The sample locations are shown on Figure 35-4. The soil samples were analyzed for PCBs, lead and magnesium.

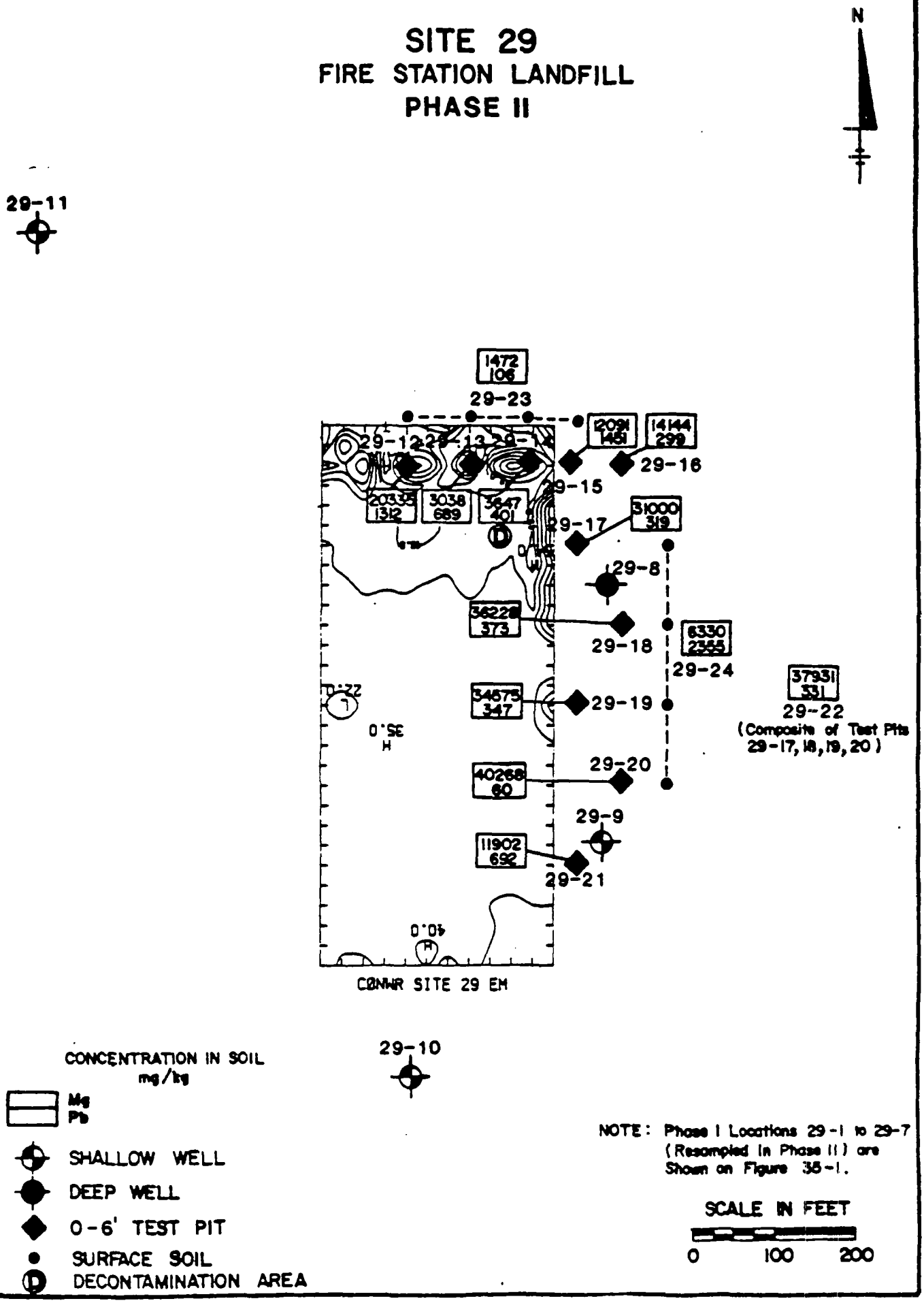
The four monitoring wells installed in Phase I were sampled and the ground water samples were analyzed for HSL CLP purgeables, pesticides, metals, and cyanide.

35.2.3 Site Hydrogeologic Characterization

35.2.3.1 Site Geology

Information obtained from the subsurface soil boring and well installation program (Borings 29-8 to 11) indicates that the site is underlain by approximately 13.5 to 28 ft. of unconsolidated sediments consisting of silty clay and clayey silt with trace sand content, overlying a light brown, medium grained sandstone bedrock. The bedrock encountered is similar to that encountered at most of the other sites. The

SITE 29
FIRE STATION LANDFILL
PHASE II



silty clay/clayey silt overburden appears to be continuous throughout the site (see Appendix B). Bedrock is also laterally continuous, as it was encountered in all four on-site borings.

35.2.3.2 Site Hydrogeology

Occurrence of Ground Water

Shallow ground water occurring beneath the site was located consistently within the silty clay/clayey silt overburden. Based on inspection of saturation in soil core samples and ground water levels in wells, ground water in this unit appears to be unconfined similar to other areas investigated. Depth to ground water was found to range 1 to 7 ft. below ground level. Ground water elevations collected during the winter and summer of 1987 (wet and dry seasons, respectively) indicate a water table fluctuation of between 0.5 and 5 ft., with water levels dropping during the summer months (Table 4-3).

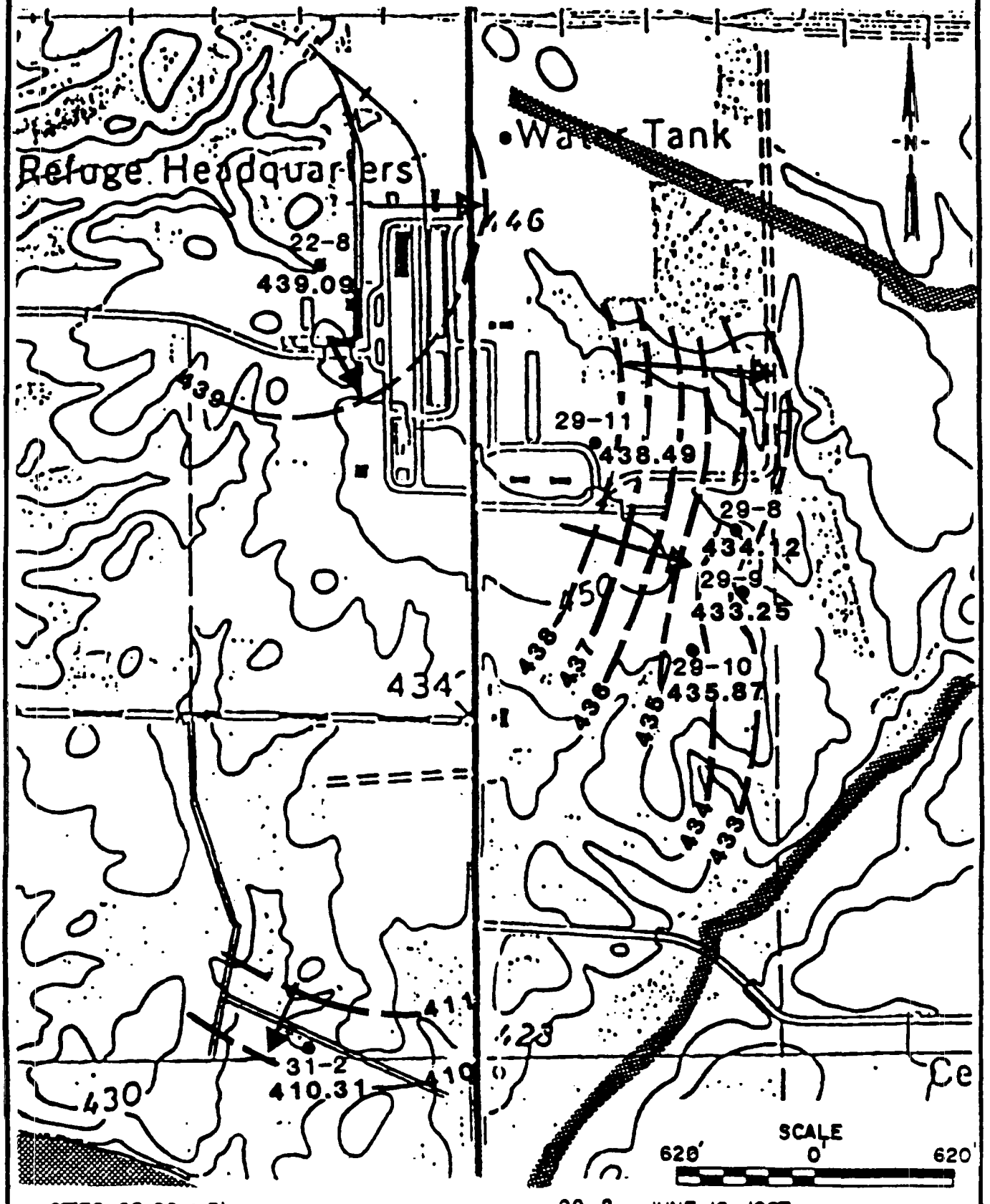
Ground Water Flow Conditions

Ground water elevations from the shallow ground water monitoring wells (29-8 to 11) were contoured and are presented on Figure 35-5. The ground water flow direction shown in this figure indicates a flow direction to the east, and towards an adjacent stream which flows southwest into Crab Orchard Lake. The average hydraulic conductivity (K) was calculated to be 1.56 ft/day. The hydraulic gradient (i) during June 18-19, 1987 was about 0.007 ft/ft as calculated from the four on-site monitoring wells. Porosity was estimated to be 0.35 (Davis and Dewiest).

A calculation was then made of the average ground water flow velocity (Vs) through the upper soil units. Using the formula given in

FIGURE 35-5

SITES 22, 29 &31 GROUNDWATER FLOW MAP



O'BRIEN & GERE ENGINEERS, INC

Section 4.2, the resultant velocity was computed to be about 0.031 ft/day or 11.32 ft/year. The flow velocity is controlled by the relatively low hydraulic gradient and hydraulic conductivity of soils occurring in this area.

35.3 Analytical Results (See Appendix I, Page 28)

35.3.1 Phase I Analytical Results:

The magnetometer and electromagnetic survey shown in Figures 35-2 and 35-3 confirmed that the northern and eastern edges of the field are the areas containing metallic debris.

The metals values reported are estimated values to be used for screening purposes (see Exhibit B). Lead concentrations in the soil ranged from 20 to 280 mg/kg, consistent with the concentrations detected at the control sites, except for sample 29-1 from the east face which contained 410 mg/kg lead. Zinc levels of 23-929 mg/kg were approximately two times the zinc levels detected at the control sites. Magnesium was also somewhat elevated throughout the site ranging from 2,430 mg/kg to 35,100 mg/kg compared to levels of 1,210-1,380 mg/kg detected at the control sites. The highest lead concentration was found in the sample with the highest magnesium concentration. Mercury was detected in six soils along the east and north at concentrations ranging from below 1 ug/kg to 5.9 ug/kg but mercury analyses were questioned due to QA/QC deficiencies and thus this parameter was reanalyzed in Phase II. Sample 29-2 with the highest FID scan (19,123 ug/kg) of all the soils was analyzed for CLP organics, but none were detected.

35.3.2 Phase II Analytical Results:

The six soils resampled for mercury contained between 23-290 ug/kg, which are higher than background for the Refuge. PCB levels in the test pit soils ranged from less than 0.40 mg/kg to 2.10 mg/kg wet weight (less than 0.049 mg/kg to 2.56 mg/kg dry weight). Magnesium and lead were detected in all of the samples ranging from 1,472 mg/kg to 40,268 mg/kg and 60 mg/kg to 2,355 mg/kg respectively (see Figure 35-4). The samples collected from the landfill contained between 3,038 and 40,268 mg/kg magnesium, and between 60 and 1,451 mg/kg lead. The composite samples collected from the north and east transects of the ditch contained 1,472 and 6,330 mg/kg magnesium, and 106 and 2,355 mg/kg lead. The higher concentrations exceeded the levels for both metals at the control site soils by more than one order of magnitude.

The ground water samples contained acetone (23-11,500 ug/L), benzene (4 ug/L, below detection limit), iron (388 - 4,000 ug/L total and less than 25 ug/L dissolved), manganese (43 - 1,790 ug/L total and 24 - 1,770 ug/L dissolved), selenium (less than 2.5-41 ug/L total and less than 14-30 ug/L dissolved), and zinc (39-140 ug/L total and 16-78 ug/L dissolved). Cyanide concentrations were below the detection level of 0.05 mg/L. The parameters which exceeded the Illinois Public Water Supply and/or Federal Drinking Water standards were iron (standards of 1,000 (State) and 300 (Federal) ug/L), manganese (State and Federal standards of 50 ug/L) and selenium (State and Federal 10 ug/L); however, dissolved metals concentrations exceeded only the standard for manganese. Iron and manganese are regulated compounds due to their objectionable taste and color in water; the levels reported at this site do not pose a risk to humans or wildlife. One sample which contained 4 ug/L benzene was

above the AWQC for human health of 1.5 ug/L. Acetone was the only other organic detected but it was also present in the QA/QC blank.

Figure 35-1 illustrates the results for mercury, magnesium and lead at the site.

35.4 Environmental Effects

35.4.1 Qualitative Assessment

35.4.1.1 Source Evaluation

The Fire Station site is an abandoned field where various waste materials including mining machinery, are alleged to have been disposed of in the past. The area is currently well vegetated with brush and tall grasses. Some exposed waste materials are present along the north and east faces of the site. There is a low area adjacent to the landfilled area which contains standing water during wet weather months.

In general, little is known regarding the nature or origin of the waste materials. There were no containerized or free phase chemical wastes located during the sampling excavations. However, an empty drum with markings indicating that it had at one time contained magnesium metal was encountered. Based on the test pit explorations, it was determined that the site contained solid waste of mixed origin. The field team did not recognize any wastes such as beverage cans and food jars which might be classified as municipal refuse. There is no information or record available regarding the amount and chemical composition of the landfilled waste materials. Therefore, this risk assessment will be based upon the residues detected.

The results of the analytical investigations are consistent with the absence of chemical wastes at this site and the possibility that it may have been used for the disposal of munitions wastes. There were no organics detected in soils at this site. There were also no PCBs or pesticides detected. The only organic chemical detected was acetone, which was detected in ground water, but this may be attributed to residual acetone on the sampling equipment following decontamination with acetone. Acetone is also commonly detected as a lab blank contaminant. The chemical components detected, including zinc, lead and magnesium, are consistent with the potential use of the area for the disposal of munitions wastes. These metals, specifically magnesium and lead, are commonly encountered at munitions waste disposal areas. On the basis of the concentrations at which magnesium and lead were encountered and the potential acute and chronic toxicity of lead, magnesium and lead were chosen as site indicators for the risk assessment. Toxicological profiles for lead and magnesium are presented in Exhibit A.

35.4.1.2 Transport Route Evaluation

Air Route

The surface of the site is generally well vegetated. There were no large areas of exposed wastes prone to erosion and dust generation encountered during the course of the field investigation. Also, there was no evidence of vehicular or foot traffic over the waste site which might lead to the erosion of the cover and the generation of airborne dusts. Because there were no measurable residues of volatile organic materials detected in the soils sampled, it can be concluded that there are no volatile wastes present on the site surface which might evaporate and be

transported via the air route. On the basis of the above considerations, it can be concluded that the air route is not functional at this location. The air transport route will not be considered further in this evaluation except as applied to exposures to dust by burrowing rodents. Neither humans or non-burrowing wildlife would receive exposures to contaminated dust particles, discussed above.

Direct Contact

As stated above, the surface of the landfilled area is covered and generally well vegetated, with few exposed waste materials. Exposed wastes encountered are limited to a few pieces of scrap metal and debris. On this basis the direct contact route at the landfilled area is considered to be non-functional. However, the direct contact route will be considered in conjunction with the surface water transport route.

Ground Water

Analyses of samples of ground water collected from monitoring wells located at the periphery of the landfilled area contained detectable residues of acetone. It has been suggested in the source characterization section that the acetone residue may be an analytical or sampling artifact. However, this cannot be substantiated. Also, manganese concentrations above State and Federal standards were detected in ground water. Therefore, the residues will be accepted as reported. On this basis, the ground water transport route will be considered to be functional.

Surface Water

It has been established that the site is currently covered by a layer of vegetation and that there are few exposed waste materials in the landfilled area. Therefore, there is a low likelihood that waste materials are being eroded from the site by the action of surface precipitation. However, the landfill is situated adjacent to a steep grade which terminates at a ditch that drains into the wet area near the site. On the basis that a steep grade and a temporary surface water body exist near the site, it can be concluded that the surface water transport pathway can function at some time by scouring materials from the top of the landfilled areas and transporting them to the wet area. It is also possible that leachate may be generated and released along the face of the grade and be transported with runoff into the ditch and adjacent wet areas. These mechanisms could establish a source of waste materials for exposures via the ingestion and direct contact routes. On this basis, the surface water transport pathway will be concluded to be functional at this location.

35.4.1.3 Receptor Evaluation

The area where the landfilled waste materials are located is a considerable distance behind the Refuge Fire Station. This area is not used for recreational or industrial purposes. This area is also not near properties used for the propagation of agricultural commodities such as meat and dairy products or grain. The primary human activity nearest to this location is related to the Fire Station. Consequently, human activity in this area is expected to be very low. On the other hand, wildlife may inhabit the area. The low wet area and adjacent wooded areas may be

inhabited by wildlife such as small mammals and birds, as well as by invertebrates and amphibians. The area may also be the habitat for deer. There are no ground water users located in the area.

35.4.1.4 Summary of Complete Exposure Pathways

Based on the above considerations, it has been established that the air and direct contact routes at the landfill surface are incomplete, based on the non-functional nature of these transport routes. One eception is noted for the air route: exposure to contaminated dusts by small rodents which could burrow into landfilled areas. The ground water route has been determined to be functional based on the detection of ground water residues. However, there are no ground water users in the area. Therefore, the ground water route can also be concluded to be incomplete.

The primary route determined to be complete is the surface water route, because of the possibility for erosion and transport of wastes towards the adjacent ditch and wet areas, where wildlife or human recreational users may encounter the materials. This transport and exposure route will be considered in the quantitative assessment.

35.4.2 Quantitative Assessment

35.4.2.1 Estimate of Release and Exposure Rates

Surface Water Mediated Direct Contact Exposures:

The surface water transport route has been determined to be complete. Transport by this route could result in the establishment of residues of landfilled wastes in the ditch adjacent to the waste site and and other surfaces at downstream locations. A direct contact scenario involving wildlife or human receptors coming into contact with waste

residues in the ditch is possible, under a variety of circumstances. However, human exposures are much less likely to occur by this scenario than would be exposures experienced by wildlife inhabiting the wet areas.

Levels of lead detected in the landfilled areas ranged from 60 to 2,355 mg/kg. One composite sediment sample from the north ditch showed a lead concentration of 106 mg/kg, while a composite sediment from the east section of the ditch contained the highest lead level detected at 2,355 mg/kg. Assuming that humans will not repeatedly visit the site, exposures to soils in the ditch will be only on an acute (one time) basis. Previous evaluations presented in this report (see Section 24.4.2) assume that a human recreational user might ingest 100 mg soil during a four hour recreational visit to any particular site. If we assume exposures to soils in the ditch can occur highest concentrations of lead detected on the site, the following exposure can be calculated:

100 mg soil/visit x 2.35 ug lead/mg soil = 235.5 ug lead/visit,

235.5 ug lead/visit x 1/70 kg body wt. = 3.36 ug/kg/visit

The above calculation shows that, for a 70 kg human, this exposure represents a single dose of 0.00336 mg/kg/visit. Compared to a reported minimum chronic toxic dose of lead to a female subject over a 6 year duration of 450 mg/kg (RTECS, 1986) and a chronic no effect level of 0.32 mg/kg/day (USEPA HEA September, 1984), the dose calculated for this site represents a non-toxic level. For a worst case of three visits/year to this site, every year during a 70 year lifetime, each resulting in exposures of this magnitude, an annual lead exposure rate of 0.027 ug/kg/day is obtained, over 10,000-fold below the chronic no effect level. On this basis, it can be concluded that acute, and even chronic exposures potentially experienced by recreators who might spend a day in

the Fire Station Landfill area would not represent unacceptable exposures, even under the scenario of repeated exposures described above.

In contrast to the limited likelihood of human exposures at this site, wildlife, such as small mammals, birds and amphibians inhabiting the area could experience chronic exposures to residues of lead and magnesium present at the site. Since wildlife would likely roam over the landfilled areas as well as the ditches, it will be assumed that such species are exposed to lead levels equal to one-half the highest concentration detected at the site or 1,177 mg/kg lead. The typical wildlife direct contact exposure estimates used in previous sections have been based on a field mouse, deer, rabbit and bird; these species will also be applicable for the wildlife assessment at this site.

Using the soil ingestion and inhalation rates presented in the wildlife risk assessment for the Job Corps Landfill (Section 24.4.2), potential receptor wildlife species at this site may be exposed to the following levels of site lead residues:

		Estimated Daily (Chronic) Intake - Lead		
	Body Weight	Inhalation Rate	Ingestion	Total
Species	(kg)	ug/kg/day	ug/kg/day	mg/kg/day
Deer	60	NA	6.3	6.3
Mallard	1	[illegible]	[illegible]	11.2
Rabbit	1.0	NA	16.8	16.8
Mouse	0.03	2.3	11.2	11.2

NOTE: See Table 24-1 (Section 24.4.2.2) for assumptions.
NA = Not Applicable

.t oug t e e a e o i e t o i toxicity data on these species of wildlife, these dosages of lead, when evaluated in comparison to the previously cited USEPA chronic no-effect level of 0.32 mg/kg-day for humans (which was generated on the basis of rat studies) indicates that

the doses calculated above may represent levels at which chronic toxicity could be experienced by wildlife.

35.4.2.2 Toxicological Assessment

The above analysis indicates a low likelihood for humans to come into contact with wastes present in the vicinity of the landfilled materials. However, even in the case where a human might encounter and ingest "worst case" amounts of eroded wastes, the dose experienced would not be likely to cause the human to experience toxic effects.

In contrast to the limited potential for the waste materials to represent an unacceptable risk to human receptors, it is possible for wildlife in the area to ingest residues of lead associated with eroded wastes at concentrations which would represent toxic effects under a chronic exposure scenario.

35.4.3 Analysis of Uncertainties

The following are the major factors which should be con- sidered when evaluating the uncertainties associated with this evaluation:

1) The waste materials present within the landfilled areas have not been comprehensively characterized. Therefore, the actual amounts, physical state and chemical composition of the waste source may not be consistent with the identities and concentrations utilized in this evaluation.

2) There is limited information available on the concentrations and distribution of waste materials present in the ditch and wet areas evaluated as part of the surface water mediated direct contact

exposure route. The concentrations assumed in the quantitative assessment may be overestimated, thereby overestimating the actual risks experienced by wildlife.

3) There are no data available on the chronic toxicity of lead to wildlife suitable for use in evaluating the residues predicted at the site studied. The chronic effect level used in this assessment to evaluate potential toxicity to wildlife has been derived for humans, on the basis of laboratory animal studies. Differences in body weight and surface area, as well as metabolic considerations and wildlife disease states, represent an unquantifiable level of uncertainty which could effect the reliability of the risk estimates regarding wildlife presented in this assessment.

35.5 Preliminary Remedial Alternatives

Based on the site investigations and the analyses of environmental effects, remedial actions aimed at reducing exposure to lead residues in soil by wildlife might be required. In addition, removal or containment of soil may be required in some areas of the landfill, particularly if additional characterization shows the presence of EP Toxic Sediments.

The contaminants detected above Refuge background levels include lead, magnesium, mercury and zinc in soil matrices. Most soil samples including surface and test pits (to a depth of 6 ft.) were taken along the limits of the fill area, as evidenced by the grade along the north and eastern edges, and thus exhibited high metals concentrations. A field transect taken approximately 50 ft. away from the grade apparently was beyond the limits of the fill along the north face, since it contained levels of magnesium and lead similar to the

range for Refuge background soils. The limits of the landfill along the eastern face may extend beyond 50 ft. from the eastern grade since the data for the east face transect sample contained levels of lead and magnesium similar to the levels found in the samples from the edge of the landfill.

The unfiltered samples of ground water contained iron, manganese, and selenium above the State Public Water Supply standards, but the metals were mostly associated with particulates and thus would not be expected to migrate readily in the groundwater table. Moreover, iron and manganese constituents were not considered to pose any health concerns at the levels detected.

Any future land uses should be evaluated prior to approval by the Refuge Management. Some of the potentially applicable remedial actions for this site are discussed below. A summary of the potential response options for this site is included in the Executive Summary (Table 2).

Limited Site Access

It may be advisable to limit human and wildlife exposure to water or soils at the Fire Station Landfill through construction of a fence around the site. Access could be controlled by the installation of locked gates to ensure that only authorized personnel enter the site. Deed restrictions might be imposed to limit future uses of the site.

Removal or Containment of Soil

One alternative to control the migration of metals offsite might involve capping the edges of the landfill with a clean, relatively impermeable cap. Additional sampling may be warranted to further define

the limits of the landfilled area and the extent of contamination in the east wetlands. EP Toxicity analyses might be conducted to determine if the lead wastes are hazardous.

Monitoring - Ground Water

Follow-up remediation efforts may include periodic sampling and analyses of the four monitoring wells for lead, magnesium, manganese, mercury, and zinc.

In addition, studies to verify the adequacy of the cleanup might begin immediately following remediation.

35.6 Conclusions and Recommendations

It can be concluded that the Fire Station Landfill site is impacted with the principal pollutant being lead. It is recommended that remedial alternatives for this site be evaluated in the FS. Potentially applicable remedial measures include removal or containment of contaminated hot spots, surface water control measures and ground water monitoring.

SECTION 36 - SITE 32, AREA 9 LANDFILL

36.1 Site Description

Area 9 is a manufacturing site on the Refuge. It was leased to Sangamo Electric Co., Capacitor Division from 1946 to 1962 and is currently leased to Olin Corporation. The area is comprised of a Building Complex and an inactive landfill located adjacent to the buildings (See Figure 36-1). Sangamo Electric manufactured various types of capacitors, utilizing aluminum, electrolytes, mica, silver, lead foil, and PCBs. Olin Corporation currently uses the buildings to manufacture explosives. Over the years, a number of other companies have occupied Area 9, according to the Refuge files, including, but not limited to, machining and metal fabricating industries, electrical components, and explosives manufacturing industries.

The Area 9 Landfill was reportedly used during the 1950s and early 1960s for disposal of wastes from capacitor manufacturing operations. The landfill was closed in 1964 (Ruelle, July 1984). When the landfill was active, wastes were burned, compacted in a swale and covered (Adams, May 24, 1984). Specific types of components that may have been disposed include some containing lead, acetate, PCBs (Aroclor 1254 and 1242), and PCB combustion products. Other possible disposed materials from capacitor manufacturing may have contained mica, silver, cyanide, aluminum hydroxide, aluminum oxide, gold, copper, zinc, hydrochloric acid, styrene, nitric acid, phosphoric acid, and borates. Other industrial wastes may include cyanides, printing inks and lead-based explosives. In addition, waste oils and debris were reportedly burned and covered with soil. The landfill was also possibly used for disposal of wastes generated in the manufacture of explosives. The landfill is locatec

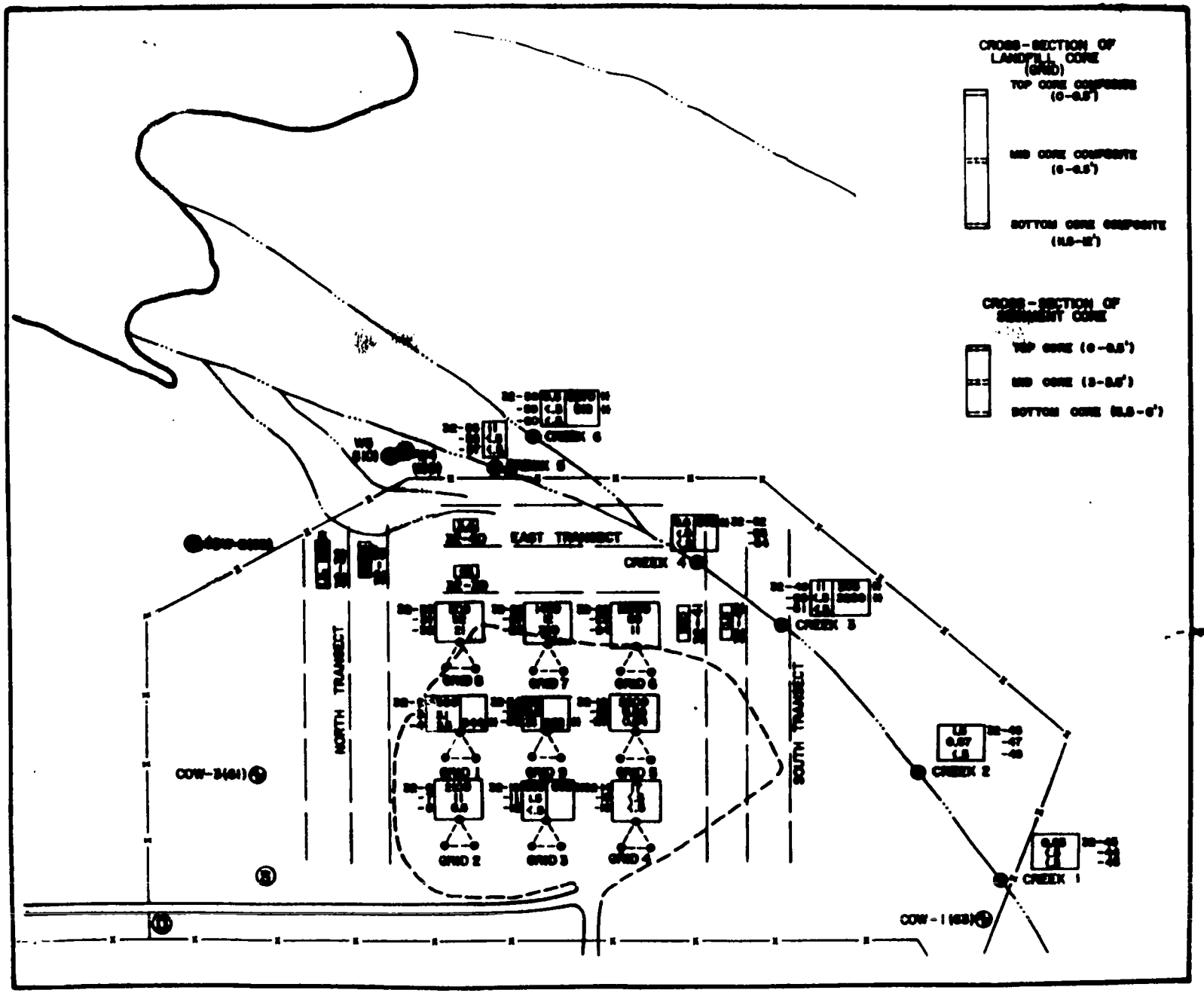


FIGURE 32-1

SITE 32
AREA 9 LANDFILL
PHASE I

LEGEND

LANDFILL BORING GRID LOCATION

SEDIMENT BORING LOCATION

PCB CONCENTRATION (mg/kg WE
LEAD CONCENTRATION (mg/kg DR
FIELDS (Phase II Results)
TOP CORE
BOTTOM CORE

LOCATION WHERE LEA CONCENTRATION EXCE BACKGROUND CONCENTR AT CONTROL SITES

DECONTAMINATION AR

WASTE STORAGE AR

NOTE: See Appendix F for Sampling L Locations.

APPROXIMATE SCALE IN FEE

0 80

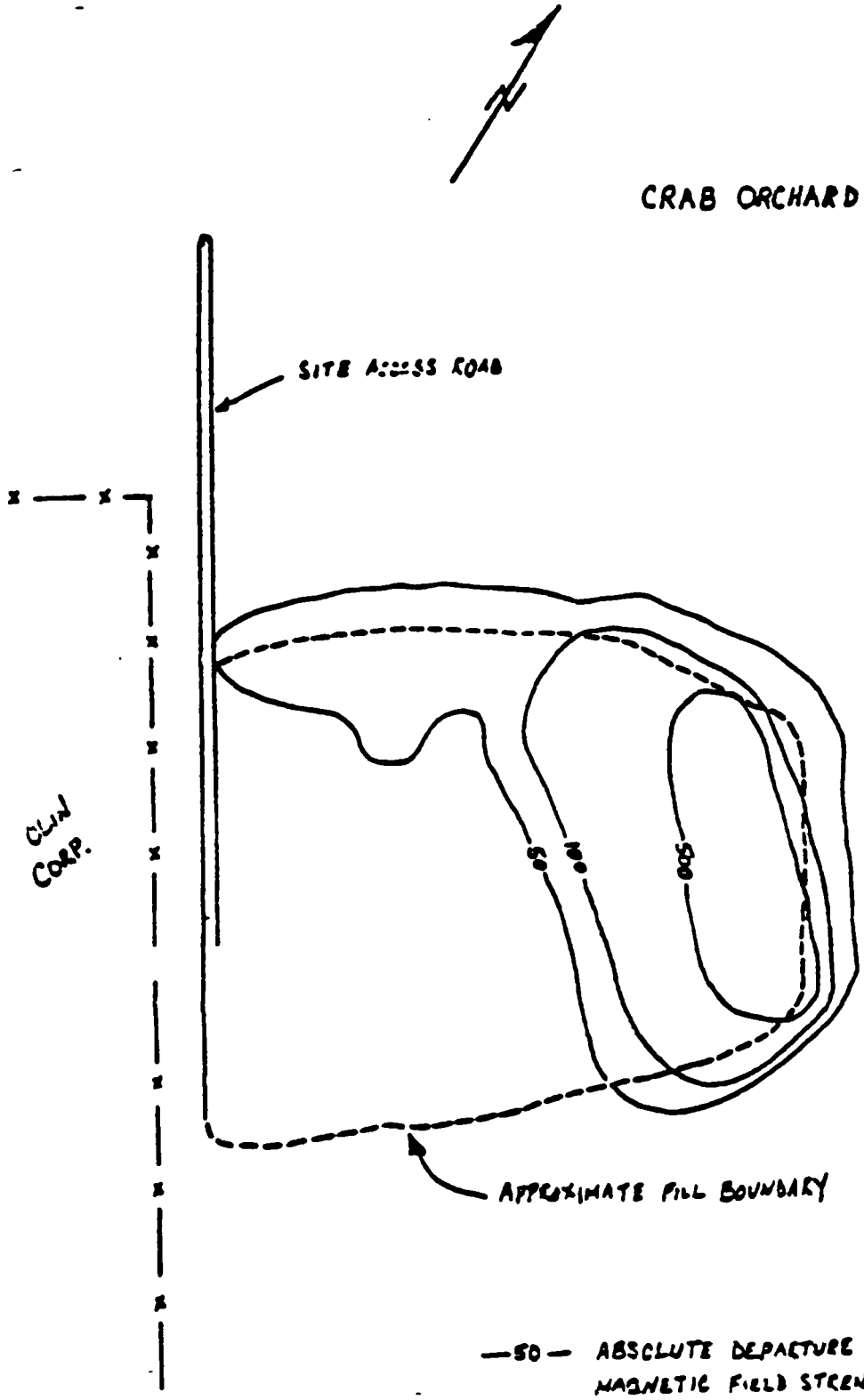
about 100 yards south of Crab Orchard Lake and 100 yards east of the building complex.

The limits of the landfill are discernible by changes in the topography and vegetation, revealing an area of approximately 2.5 acres with an estimated fill thickness of 8 to 10 feet in the middle and 6 feet at the edges. The landfill is covered by a thick growth of tall grass except where waste materials are exposed. The volume of the landfill is estimated to be from 16,000 to 35,000 cubic yards. Materials visible on the surface appear to be electrical components consisting of small capacitors, capacitor parts, chunks of a golden resin, and a number of 3-inch steel cuplike pieces.

A magnetometer survey conducted by DOI suggested that the majority of wastes are buried along the eastern and northern edges of the landfill. (See Figure 36-2). Runoff from the landfill can drain into an intermittent creek and into Crab Orchard Lake.

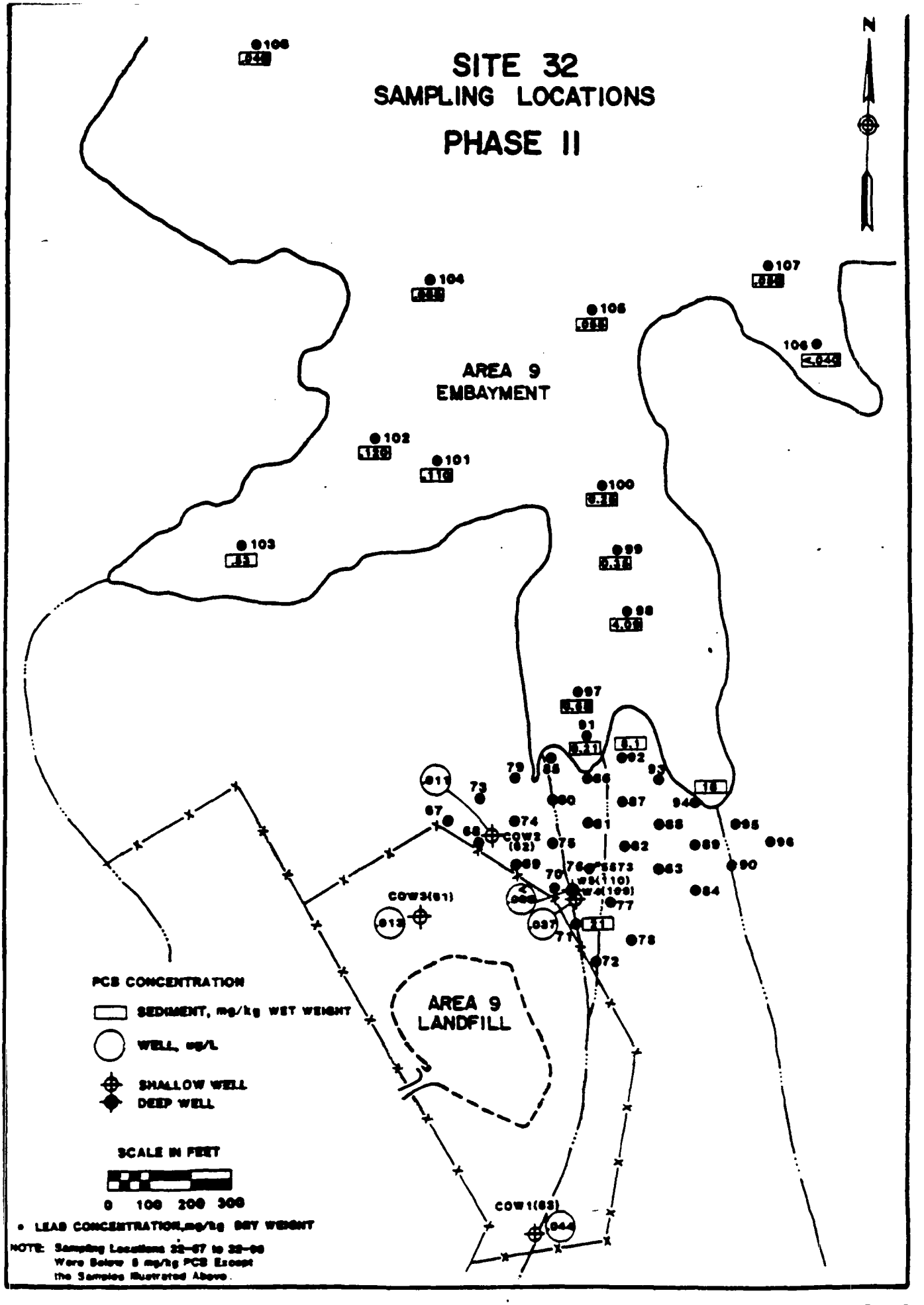
Three ground water monitoring wells were installed in the vicinity of the landfill in late 1983 prior to the Phase I investigation (Gifford, 1984). (See Figure 36-1, Wells COW1-COW3). The wells were installed in silty sand to depths of 29, 35.5, and 30.5 ft., and screened the bottom 5-10 ft. of the boring. Boring logs from previous investigations by Illinois EPA at the Area 9 landfill reveal that the site is underlain by a minimum of 7.5 feet of clay. The silty clay is reported to be upwards of 30 feet thick north of the landfill and 25 feet thick south of the site. Groundwater, at locations around the landfill, was reported to be 2 to 24 feet below the surface during different time periods (Gifford, 1984).

SITE 32
MAGNETOMETER SURVEY



SOURCE:
Byram, Scott. Memo to File. Crab Orchard National Wildlife Refuge– Sangamo Dump. TDD No. R5-8308-6. September 21,1983.

O'BRIEN & GERE ENGINEERS INC



O'BRIEN & GERE
ENGINEERS INC

36.2 Site Investigations

36.2.1 Phase I Site Investigations:

Nine compositing stations were established within the landfill area for collection of soil samples. (See Figure 36-1). Samples were obtained from one-foot intervals from the surface to a depth of 12 feet.

The exact boundaries of the landfill are unknown because contaminants could have washed from elevated portions of the landfill onto the lower surrounding area. To identify the extent of contaminant transport from the landfill to surrounding areas, surface soil subsamples were collected at 3-foot intervals along each of six transect lines, two each on the east, south and west side of the landfill as shown on Figure 36-1. The soil composites at each depth (top, mid, and bottom cores) were analyzed for PCBs, indicator compounds and PCDD/PCDF screening. A composite for each grid location (0-12 ft depth) was analyzed for the full priority pollutants screen.

Grab sediment samples (0-1 ft depth) were also collected from six locations along the intermittent creek adjacent to the landfill. The sediments were screened for priority pollutants, metals, explosives, and cyanide.

36.2.2 Phase II Site Investigations:

Thirty surface soil samples were collected from the lowland area northeast of the landfill. These samples were collected from a 5 sample by 6 sample grid with grid points approximately 100 ft. apart. Twelve sediment samples were also collected from the lake embayment downstream (north) of the landfill. (See Figure 36-3). The soil and sediment samples were analyzed for PCBs and lead. Five of these soils were analyzed for full CLP organics also.

Twenty-four surface and nine core soils collected in Phase I were reanalyzed in Phase II for mercury, chromium, and lead. Mercury was reanalyzed because the Phase I results were questionable due to QA/QC deficiencies.

Three additional monitoring wells were installed during Phase II (Figure 36-3). Well 32-63 was installed south of the landfill near well COW-1, one of the three wells installed in 1983. COW-1 could not be located. Further information describing the wells installed prior to the RI/FS is provided in Section 36-1. Wells 32-63 and 32-109 in Phase II were installed in silty and silty clay soils to a depth of 15 feet and screened from 10 to 15 feet. Well 32-109 was nested with well 32-110. Well 32-110, an artesian well, was installed on top of bedrock at 82.5 feet and screened within sandy soils from 77.5 to 82.5 feet. The five monitoring wells (including the wells installed prior to this RI) were sampled and ground water samples were analyzed for full CLP organics, metals, low level nitrosamines and cyanide.

In addition to the soil and water samples, several capacitor parts were found either buried or partially buried along the drainage ditch northeast of the Building Complex. The capacitor casings and contents were analyzed for PCBs and lead, as were several soil samples collected from the area where they were found.

36.2.3 Site Hydrogeological Characterization

36.2.3.1 Site Geology

Information obtained from the subsurface soil boring and well installation program indicate that the site is underlain by approximately 77 feet of unconsolidated sediments overlying sandstone bedrock. The

particular sequence of soil units encountered in the deep well No. 32-110 consisted of alternating coarse to fine sand and silty clay units 5-25 feet thick. Similar to other sites, the bedrock was overlain by a sand layer. This area contained, however, additional sand layers not found in other sites. Since no other deep borings were drilled in this area, the lateral extent of the sand layers is not known. Typical of the area, the bedrock encountered at a depth of 77.5 feet consisted of a gray, medium grained, sandstone with the upper 1-2 feet highly weathered.

36.2.3.2 Site Hydrogeology

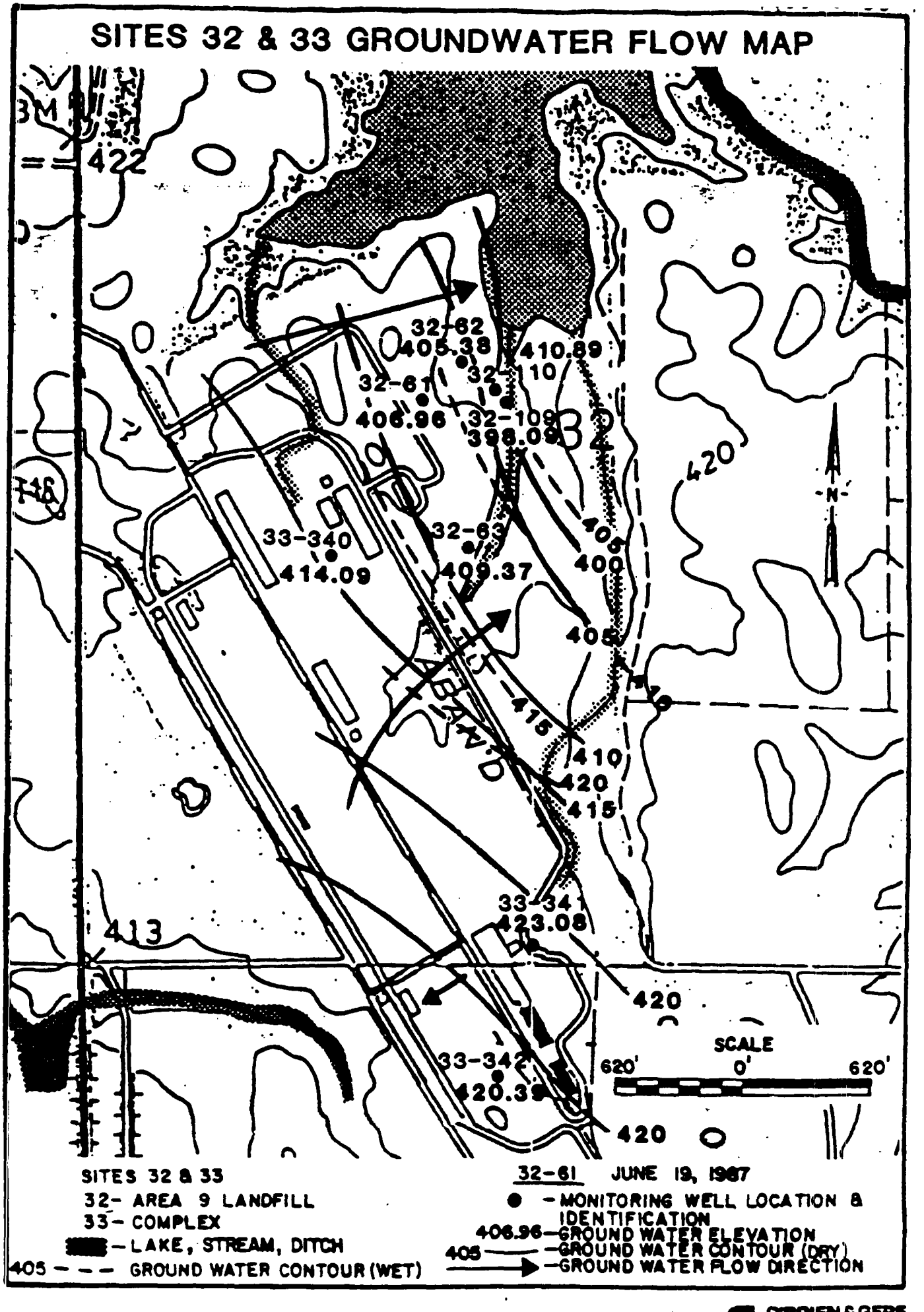
Occurrence of Ground Water

Shallow ground water occurring beneath the site was located about 5-12 feet below ground surface in June 1987 within upper silty clay and fine sand layers. Ground water levels were higher during the wet season (Dec. 1986) by about 3-10 feet. Ground water in these units appears to be unconfined similar to other areas investigated. Ground water occurring within lower portions of the soil sequence on top of bedrock was found to be confined and yielded a continuous flow at ground surface of about 5 gal/min. Ground water within underlying bedrock was not investigated.

Ground Water Flow Conditions

Ground water elevations from the shallow monitoring wells were contoured and are presented on Figure 36-4. The ground water flow direction exhibited in this figure indicates a flow to the northeast toward Crab Orchard Lake. The hydraulic gradient of flow (i) during June 18-19, 1987 was about 0.015 ft/ft. The average hydraulic conductivity

SITES 32 & 33 GROUNDWATER FLOW MAP



O'BRIEN & GERE ENGINEERS, INC.

(k) was calculated to be about 0.56 ft/day from the five shallow monitoring wells. The porosity (n) was estimated to be 0.35 (Davis and Dewiest).

A calculation was then made of the average ground water flow velocity (Vs) through the upper soil units. Using the formula given in Section 4.2, the resultant velocity was calculated to be about 0.024 feet/day or 8.7 feet/year. The flow velocity is controlled by the relatively low hydraulic gradient and hydraulic conductivity occurring in this area.

An upward vertical flow potential was identified from the unconsolidated aquifer screened by the deep well 32-110 into the upper water table. This phenomenon indicates discharge of deep ground water toward adjacent Crab Orchard Lake, similar to the shallow ground water.

36.3 Analytical Results (See Appendix I, Page 29)

36.3.1 Phase I Analytical Results:

Figure 36-1 presents the results of PCB analyses within the Area 9 landfill. The PCBs are generally present above the TSCA criterion of 50 mg/kg only on the surface (0-1 ft) except for the eastern edge of the landfill area. PCBs above 50 mg/kg were observed at the 6 ft depth in the northeast (grids no. 6 and 8). Grid number 7 was the only station where soils contained PCBs above 50 mg/kg at the 12 ft depth. (See Figure 36-1).

PCB levels in the soil samples from the transects on the north, south and east sides of the fill area were above 50 mg/kg at the surface, immediately adjacent to the landfill but were below 50 mg/kg on the adjacent transect on all sides. (See Figure 36-1). Sediments in the

drainage channel upstream and downstream of the landfill contained wet weight PCB concentrations of 0.86-1.6 mg/kg, and 0.8-11 mg/kg, respectively, while PCBs were undetected (less than 0.5 mg/kg) for all the 3 ft. and 6 ft. depths of the downstream sediments.

Lead concentrations ranged from 11 to 29 mg/kg in 0-3 ft. composite sediments from the creeks around the landfill, and from 0.9 to 25 mg/kg in the 0-12 ft. composite soil samples from the landfill that were analyzed for lead. However, the metals concentrations reported are not supported by QA/QC (see Exhibit B). Selected Phase I grab soil samples which had not been previously analyzed for lead were resubmitted during the Phase II program for analysis of lead, mercury, and chromium. The reanalyses were considered necessary due to the uncertainty inherent in the Phase I analyses, and also to better quantify the levels present in the grabs at specific depths. Nine out of 32 soil samples reanalyzed showed lead concentrations outside the common range of 2-200 mg/kg in soils (Lindsay, 1979). Figure 36-1 shows the lead concentrations for these 9 samples ranging from 205 to 8,570 mg/kg.

The remaining samples showed lead levels similar to the Refuge control sites. Figure 36-1 shows the results for lead from the Phase II reanalyses. Further discussions of the metals analyses are given in the next section, Phase II Analytical Results. Full analyses of organic priority pollutants were conducted on each of the nine boring composites (0-12 ft depth). The concentrations of heavy metals and other contaminants present in the composite samples from the landfill did not differ significantly from those at the control sites.

The results for PCDD and PCDF isomers in soils are presented on page 30A of Appendix I. A separate data listing is included to illustrate

the actual dioxin/furan to PCB ratios compared to the ratios which would be expected based on the PCB concentration for that sample. The purpose of calculating this ratio is to determine if dioxins/dibenzofurans are elevated due to burning products of PCBs. If this were the case, then the dioxin/furan to PCB ratios would be higher than the expected normal ratios for PCBs. The Table lists a concentration for each peak detected in the scan (Isomer No.) and a total concentration for each compound. This total concentration was compared to the average PCB concentration detected in the same sample to develop a ratio of PCDD or PCDF to PCBs. An average PCB concentrations was used because the analyses were conducted in duplicate (one reported by ETC Laboratories and one reported by OBG Laboratories).

Based on studies conducted by T. Sawyer and S. Safe et.al.(1982, 1984, 1985) relative to PCB isomers and congeners, the expected fraction of PCDD and PCDF isomers associated with a measured PCB concentration can be calculated. The 'equivalent' fraction of dioxins and furans is determined by a conversion factor or Keq value. The Keq value is based on 1.00E-05 for 2,3,7,8-TCDD.

The results for Area 9 Landfill showed positive detections above the ratios which would be expected for TCDF, PCDF, and OCDD compounds. TCDF levels ranged from 0.14 to 26.3 ug/kg in 7 out of 12 samples. PCDF was detected in one sample at 0.34 ug/kg. OCDD levels ranged between 0.6 and 20.6 ug/kg in 9 samples. There was no specific pattern discernable in the distribution between dioxins and furans.

36.3.2 Phase II Analytical Results

PCB concentrations in surface samples from the lowland area downgradient of the landfill were all below 5 mg/kg with the exception of three of the thirty samples taken. These excursions were noted at Locations 32-71, 32-92, and 32-94, with 21,8.1, and 18 mg/kg wet weight, respectively. (See Figure 36-3).

PCB concentrations in sediment samples from the lake embayment area near the landfill showed a maximum concentration of 4.09 mg/kg, while all other lake sediments (locations 32-97 through 32-108) contained PCB concentrations below 0.65 mg/kg. (See Figure 36-3).

The five ground water monitoring wells sampled during Phase II all exhibited PCB concentrations below 0.045 ug/L. All downgradient ground water concentrations (less than 0.005 to 0.037 ug/L) were less than the upgradient concentration of 0.044 ug/L. There are no state or federal standards for PCBs in water. However, PCBs in four of five monitoring wells exceeded the Ambient Water Quality Criteria for human health. PCB concentrations detected in ground water are likely associated with suspended particulates in the well, since these compounds have a high affinity for and adsorb tightly to the silty clay soils at the site. The presence of PCBs in suspended solids in the wells may result from contaminated surface sediment introduced during installation of the well, natural fluctuations in water levels causing particulates from the upper soil layers to be washed to the lower levels where the well is screened, or by adsorption of soluble PCBs onto the suspended matter already present in the well. Well 32-63 was found to contain 92 mg/L of chromium, which exceeds the Illinois Public Water Supply Standard and Federal MCL of 50 mg/L. However, the dissolved chromium concentration of 1.2 ug/L for this sample was within all applicable standards.

Acetone and methylene chloride were detected but were also present as contaminants in the QA/QC blanks. All other concentrations were within these standards. Of the 30 Phase II soil samples from the low lying area northeast of the landfill that were analyzed for lead, only one sample (32-76) contained lead concentrations significantly in excess of the concentrations detected at the control sites, with 5573 mg/kg (See Location 32-76, Figure 36-3). All other concentrations were well below the background level of 200 mg/kg. The Phase I soils from the landfill reanalyzed under Phase II generally contained concentrations within the refuge background levels for chromium.

Mercury levels ranged from less than 0.023 to 0.035 mg/kg, which are similar to the concentrations found at the control sites. Lead concentrations in the landfill were above the typical range of 2-200 ppm in soils (Lindsay, 1979) for nine out of 32 grab soils that were reanalyzed in Phase II. The lead results for selected samples which ranged from 205 to 8,270 mg/kg are shown in Figure 36-1. The lead concentrations in the remaining samples were similar to the levels detected at the control sites. Only traces of organic compounds other than PCBs were detected at the landfill.

Acetone, methylene chloride, di-n-buthylphthalate and isophorone were detected but these were also present in the QA/QC blanks. The analytical results for the capacitors found at Area 9 showed positive detections for PCB Aroclor 1254 for the cylindrical and square capacitors. (See Page 30B of Appendix I). A composite of five smaller capacitors tested positive for PCB Aroclor 1242. Some capacitors did not contain detectable concentrations of PCBs. The soil samples collected from the area where the capacitors were found also contained PCB Aroclor 1254 at

concentrations between 1,100 and 88,000 mg/kg wet weight. The capacitor casings were also analyzed for lead and were found to contain between 156 and 20,000 mg/kg wet weight of lead. The soil samples contained lead ranging from 1,810 to 20,500 mg/kg wet weight (1,880 to 37,600 mg/kg dry weight).

36.4 Environmental Effects

36.4.1 Qualitative Assessment

36.4.1.1 Source Evaluation

The results of the site investigations, as described in the preceding sections, determined that the Area 9 Landfill was used for disposal of wastes arising from the manufacture of capacitors. A variety of waste materials were disposed of at the site, including lead and PCBs. Most of the wastes are situated on the western and northern boundaries of the landfill. Runoff from the landfill can reach Crab Orchard Lake via intermittent creeks. The results of the Phase I and Phase II investigations indicate that PCBs are the major contaminants identified within the landfill site itself, while lead was the principal contaminant found in the lowland area below the landfill.

PCB levels were generally highest in the northern and western sectors of the landfill, with values in the 0 to 1 foot cores ranging from 2,100 mg/kg up to 13,000 mg/kg (Figure 36-1). Subsurface contaminant levels were considerably lower. Sediments taken from intermittent creeks draining the landfill showed low-level PCB contamination (0.8 to 11 mg/kg in six creeks sampled). Several soil samples from the landfill and some off-site creek sediments reanalyzed in Phase II contained elevated

levels of lead, ranging from 205 to 8,270 mg/kg, although no pattern of contamination was evident.

Isomers of tetrachlorodibenzofurans (TCDF) ranging from 12 to 28 ug/kg were detected in three soil samples at this site. Detection of polychlorinated dibenzo/dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) present a cause for concern due to the wide range of acute and chronic toxicity observed for one member of these classes; 2,3,7,8-tetrachloro- dibenzodioxin (2,3,7,8-TCDD). 2,3,7,8 - TCDD is highly toxic under acute exposure, and produces a number of chronic disorders including but not limited to immunotoxicity, teraogenicity and reproductive toxicity, and it is a suspect human carcinogen. There are very strict molecular requirements for production of toxic effects by these chemicals at a biochemical level, due to required interactions with receptor molecules in target tissues. 2,3,7,8-TCDD, and to a lesser extent 2,3,7,8-TCDF appear to have the optimum spatial and electronic requirements for toxic potential of all the isomers in these two classes of chemicals. However, there is no evidence at present that 2,3,7,8-TCDF or other isomers are carcinogenic in either animals or humans. Removal of chlorine or substitution at other ring positions greatly diminishes observed acute and chronic toxic effects relative to 2,3,7,8-TCDD. Addition of chlorine to PCDDs and PCDFs also decreases toxicity. Thus, a mixture of hexachlorodibenzodiozins was shown to be only a weak liver carcinogen in rodents while octachloro-dibenzodiozin is non-carcinogenic. In view of detection of a TCDD in only one site sample, lack of information on the carcinogenicity of the TCDFs and on the isomeric composition of the site TCDI and TCDF, quantitative risk estimates for these site contaminants are inappropriate at present.

Chromium was found in one groundwater sample in excess of Illinois Public Water Supply Standards and the Federal MCL. However, the dissolved chromium concentration for this sample was within all applicable standards. Trace PCB concentrations below 0.045 ug/L were detected in the ground water samples. Based on this analytical survey and the non-threshold and threshold toxicity of these chemicals, PCBs and lead were chosen to serve as site-indicator contaminants for the purpose of this risk assessment. The physicochemical and toxicological properties of PCBs and lead are summarized in Exhibit A. Both PCBs and lead adsorb tightly to the silty and silty clay soils observed at this site (see Section 36.2). This property is the dominant factor determining the environmental transport and fate of these compounds. The measured soil hydraulic conductivity at this site ranges from 4.0×10^{-6} to 2.32×10^{-6} ft/sec. The transport of PCBs and lead are significantly retarded by this type of soil.

36.4.1.2 Transport Route Evaluation

a) Air: Exposures to PCBs in the vapor state as a result of volatilization from high residue levels in soil were discussed in Section 24.4.2.1, and were determined to contribute approximately 8 percent of the total airborne exposure. In addition, the existence of exposed soil containing tightly adsorbed contaminants in some areas of the site, dusts generated by wind erosion, or the activities of endemic wildlife constitute a functional route for conveying PCBs and lead to on/off-site locations via the air route for subsequent exposures by receptors in those areas.

b) Direct Contact: Due to the presence of site indicator contaminants in exposed wastes, soils, and sediments in the area, exposures by the direct contact route are possible.

c) Surface Water: One main creek northeast of the landfill serves to drain the site and Crab Orchard Lake is 100 yards downgradient of the landfill. Furthermore, sediment analyses from the creeks showed the presence of both PCBs and lead. Therefore, the surface water transport route is considered functional via precipitation-initiated runoff events which convey soil- and sediment-bound site contaminants towards Crab Orchard Lake.

d) Ground Water: Estimates of contaminant transport via groundwater were prepared for PCBs and lead. Using the groundwater flow velocity calculated in Section 36.2.3 (0.024 ft/day) and an average PCB concentration of 0.020 ug/L for the three shallow downgradient monitoring wells, a worst case estimate of 0.20 mg/day PCBs could potentially reach the lake. This is based on a plume depth of 30 ft and width of 500 ft, since, based on the analysis of one deep well (32-110) which was screened at a depth below 30 ft., the lower aquifer did not contain detectable contaminants. The same values were used with an average dissolved lead concentration of 1.8 ug/L, resulting in an average mass loading below 18 mg/day lead. However, the actual loadings to the lake will be less than this value since several interfering mechanisms take place between the well point and the water discharge to the lake. PCBs and lead exhibit particularly strong affinities for soil and suspended particulates, and would not be expected to migrate significantly either vertically or horizontally. A significant percentage of PCB residues, if these were actually

present in ground water, will be removed by soils; dissolved lead species may also precipitate by interaction with soil matter to reduce the levels reaching the lake. Furthermore, the contaminants will be diluted by several orders of magnitude upon reaching the lake, and some persistent residue levels may adsorb to suspended solids in the water column and eventually settle.

Based on the detection of minimal site contaminants in groundwater, limited transport as demonstrated above, and no ground water uses at this site, this exposure route is incomplete and will not be considered further in the risk assessment.

36.4.1.3 Receptor Evaluation

Human

The Area 9 Landfill is situated in a non-populated area adjacent to an industrial facility, and is fenced. Therefore, the only potential human receptors would include facility employees, site trespassers possibly including children, and (given the closeness of the site to Crab Orchard Lake) occasional recreational users of the Refuge. The number of human receptors is very low and exposure will be of a transient, non-chronic nature. Specific scenarios for human exposure to site indicator contaminants will be developed in the following sections. The transport route evaluation identified three functional transport mechanisms: the air route, the direct contact route, and the surface water transport route. Exposures will generally occur only in the vicinity of the landfill, with the possible exception of downgradient creeks and lowlands near the lake, and consumption of fish taken from the lake. The following are the most likely human exposure scenarios for the functional transport routes.

a) Direct Contact: The most probable human exposure scenario would be exposure to site indicators via direct contact with surface residues at the landfill and to sediments in the several intermittent creeks that drain the landfill. These exposures will be on a one time- or very limited basis and of a short term nature. The small number of Olin facility employees constitute one group of potential receptors. Trespassers gaining access to the fenced landfill area or hikers traversing contaminated creek sediments may also be exposed to contaminants via direct contact. Although the landfill is generally well vegetated, there are areas of exposed wastes and soils that could serve as a potential source of exposure. The most likely mode of entry of contaminants into the body would be incidental ingestion of soil-bound residues adhering to the skin, clothing, or shoes, acquired by direct contact with exposed wastes.

b) Air Route: Dusts generated by wind erosion or foot traffic over exposed waste areas constitute the major source and mechanism for exposure via the air route since the site contaminants are soil-bound and non-volatile. As with the direct contact pathway, the receptors include facility employees and other passersby who may breathe contaminated dusts while traversing the landfill. These exposures will also be acute in nature.

c) Surface Water Route: Monitoring of sediments of intermittent creeks draining the landfill indicate that low concentrations of site contaminants have migrated from the landfill, presumably by surface water runoff. The presence of PCB residues in sediments of Crab Orchard Lake near the Area 9 Landfill site (Section 2.6.1) suggests that transport of contaminants to the lake is occurring. Therefore,

some degree of exposure is possible from the consumption of fish taken from the vicinity of the source which have accumulated site residues in edible tissues.

d) As presented in the preceding section, transport of site contaminants to Crab Orchard Lake sediments presents a functional human exposure pathway via ingestion of residues accumulated in fish.

Wildlife

Due to the industrial nature of this site, the diversity and abundance of habitat may be relatively less than at other, less restricted areas of the Refuge. Nevertheless, the vegetated landfill provides adequate habitat for invertebrates and small vertebrates, and the nearness of Crab Orchard Lake creates the opportunity for exposures to aquatic populations. Occassionally, deer have been seen within the landfill area; however, these animals have not shown any impact from PCBs or lead at the site based on studies summarized in Section 2.7.

a) Direct Contact: Wildlife inhabiting the landfill such as invertebrates and small burrowing rodents will receive both acute and chronic direct contact exposures to site contaminants bound to soil dusts during burrowing activities. Exposed waste areas present a functional direct contact route exposure path for birds while feeding, ingesting grit, and dusting. Subsequent ingestion of soil-bound residues while preening or grooming is the principal means of entry into the body.

b) Air Route: Inhalation exposures of wildlife to dust-bound site contaminants will follow the direct contact scenario described above. Exposures to PCBs due to their volatilization from soils were evaluated in Section 24.4.2.1, and also contribute to this exposure

route, especially for small rodents which would be in intimate contact with soils during extended periods (e.g. within a burrow).

c) Surface Water: Transport of site contaminants to Crab Orchard Lake from the landfill via runoff of sediments creates a functional chronic exposure pathway for aquatic organisms in the bay area of Crab Orchard Lake. Exposures will be relatively greatest for benthic invertebrates and bottom-feeding fishes such as catfish.

d) Ingestion: Implied in all three wildlife exposure pathways discussed above is the ingestion of site contaminants via soils, dusts, sediments, and prey consumed in various activities. In addition, herbivores may consume contaminated dusts on seeds and vegetation and fish, birds (i.e. ducks, herons) and other aquatic organisms may inadvertently ingest contaminant-bearing sediments while feeding. Site contaminants, especially PCBs, are prone to accumulate in aquatic food chains, adding to the importance of the ingestion route of exposure.

36.4.2 Quantitative Assessment

Because of the relatively greater magnitude of PCB residues at this site compared to lead, and because of potential for non-threshold toxicity presented, the quantitative assessment will focus primarily on PCBs as the site contaminants of greatest concern. Exposure to lead residues will be addressed due to the presence of lead in non-fenced areas of the site.

36.4.2.1 Estimates of Release and Exposure Rates

Estimates of Exposures by Direct Contact

The qualitative assessment for the Area 9 Landfill has determined that direct contact represents a functional exposure pathway for humans and wildlife. However, PCBs and lead are tightly bound to soils and sediments, and are poorly absorbed through the skin. Therefore, dermal absorption of contaminants is not expected. The scenario consists, instead, of ingestion of bound residues picked up through direct contact with soils and sediments. Therefore, the contribution of this route of exposure will be addressed in the section below on ingestion exposures.

Estimates of Airborne Exposures

The qualitative portion of this assessment has established that the air pathway represents a complete exposure route. The pathway consists of breathing of contaminated dusts at the landfill site by occasional human activities (visits by employees of the adjacent Olin facility, trespassers, etc.), and by burrowing and dusting activities of wildlife. The general approach and assumptions used to estimate airborne human and wildlife exposures is given in Section 24.4.2.1 of this report. Using this approach for a four hour excursion by a facility employee or trespasser in a sector of the landfill containing exposed wastes, and assuming a mean surface PCB soil level of 3,200 mg/kg, a total exposure to PCB of 166 ug/day, or 2.4 ug/kg body weight is obtained as an inhalation exposure rate. Assuming three such visits to the site per year, a chronic inhalation rate of 0.02 ug/kg/day is derived. It should be realized that such a scenario does not technically define a chronic exposure, and is essentially invalid on toxicological grounds. However, regulatory agencies currently suggest this approach to estimate the "worst case". The acute worst-case inhalation exposure level of 2.4 ug/kg is far lower than would create any concern for acute toxicity. Using a representative intermediate lead soil

concentration of 4,000 mg/kg, a one day exposure of 3.0 ug/kg is estimated under the same worst case conditions. The contribution of inhaled residues to total acute intake is discussed in Section 36.4.2.2. True chronic inhalation exposures are likely, however, for small burrowing mammals at the site. Using a breathing rate value of 0.006 m3/hour for a 30 g mouse (U.S. EPA 1985) and creation of a 10 mg/m3 dust containing 3,200 mg/kg of PCBs during 1 hour of daily burrowing, a daily chronic exposure of 6.4 ug/kg/day is obtained for inhalation of dusts. In addition, although PCBs exhibit a low vapor pressure it will be assumed that at the concentrations found in surface soils the air is saturated with PCB vapors. This assumption would not likely hold at low levels of PCBs in soil due to the adsorptive forces of these compounds with soil. Small rodents living in burrow areas at the site would inhale these saturated vapors. The assumptions used to derive an exposure level for PCB vapors were detailed in Section 24.4.2.2. Applying a similar rationale for rodents living at the Area 9 Landfill, mice might inhale 1.09 mg/kg/day if exposed to saturated vapors during 16 hours inside a burrow. The total inhalation exposure from PCBs is thus (1.09 + 0.0064) = 1.1 mg/kg/day.

Exposure to lead, on the other hand, would be mainly through inhalation of dusts during burrowing. For lead concentrations in soil/sediment at 4,000 mg/kg, the exposure rate would be 5.6 ug/kg/day. The significance of these exposures is discussed later in Section 36.4.2.2.

Estimates of Ground Water Exposures

It has previously been determined that the groundwater exposure pathway is incomplete at the Area 9 Landfill and therefore will not be considered quantitatively.

Estimates of Surface Water Exposures

In view of a functional transport mechanism for conveying site contaminants to Crab Orchard Lake via runoff events, the surface water pathway is complete. The actual mechanism of exposure will be via ingestion of contaminants accumulated from residues present in sediments of a bay of the lake adjacent to the landfill. Therefore, exposures by this route will be discussed in the following section on ingestion exposure.

Estimates of Ingestion Exposures

Ingestion exposure of site contaminants at the Area 9 Landfill and contiguous sites has two components: ingestion of soil-bound residues acquired by direct contact with waste materials, and bioconcentration and foodchain accumulation of PCBs in a nearby bay of Crab Orchard Lake, with potential exposures to humans and terrestrial wildlife consuming contaminated aquatic organisms. The basic approach and assumptions used to estimate human and wildlife exposures by direct contact and ingestion of contaminated soils have been discussed in Section 24.4.2.1, the quantitative assessment for the Job Corps site. Using the worst case assumption that an individual ingests 100 mg of soil as a result of an excursion into an exposed waste area of the Area 9 Landfill, and that the mean surface level of PCBs is 3,200 mg/kg, an ingestion rate of 4.6 ug/kg/visit is estimated for a 70 kg adult. Using the same assumptions and a exposed surface soil lead concentration of 4,000 mg/kg, an acute ingestion of lead is estimated at 5.7 ug/kg/visit. Wildlife PCB intake rates from ingestion of contaminated soil at the site during feeding or grooming are detailed in Section 36.4.2.2, Quantitative Assessment.

Estimates of intakes to piscivorous mammals such as mink and otter, which could be exposed to residues of PCBs from ingestion of fish in the area of the Area 9 Embayment, are presented in Section 38.4, Environmental Effects for Crab Orchard Lake. Exposures to fish-eating birds (bald-eagle, osprey, duck) is also addressed in Section 38.4.

Ingestion of residues found in the water of the Area 9 Embayment are possible for terrestrial mammals. Water intake rates for such species will be assumed to be 10 percent of body weight for hervibores and 30 percent of body weight for carnivores. The PCB concentration in a water column sample taken from the lake, just outside of the embayment was 19 ng/L (ppt). Assuming this concentration is representative of the bay area water, it is estimated that the three hervibores evaluated (mallard duck, rabbit and mouse) each receive an exposure of 1.9 ng/kg/day.

36.4.2.2 Quantitative Risk Assessment

Human Risks

Human exposure at the Area 9 Landfill will be extremely limited. With the installation of a chain link fence around the landfill in 1984, the direct contact and airborne pathways described above may very well be incomplete. Visits by Olin employees would be presumably authorized with appropriate protective equipment used, and excursions by hunters and hikers are eliminated. Only intentional trespassing onto the landfill would provide a complete pathway. Human exposure estimates for airborne dust-bound PCB residues (2.4 ug/kg/day) and direct contact ingestion of soil-bound residues of PCBs (4.6 ug/kg/visit) provide a total estimate of 7.0 ug/kg/visit for this scenario. This intake is far lower than any that would be cause for concern from acute PCB toxicity. Chronic PCB

exposures are not possible for this site and it would be inappropriate to derive estimates for such.

U.S. EPA (1987) developed a drinking water health advisory, however, for short term exposure to PCBs (Aroclor 1254) in drinking water. Selecting a representative no observed adverse effect level of 1 mg/kg/day based on effects on liver weight in rats exposed for seven days, an acceptable short term, 10 day, exposure level in humans of 0.7 mg/day, or 10 ug/kg/day for a 70 kg adult, was derived. Apportioning the upper level acute exposure estimate for a trespasser within the Area 9 landfill over a period of 10 days, results in a daily intake of 0.26 ug/kg/day, a level within the short term acceptable exposure limit. This estimated intake level assumes that a trespasser may be exposed as a result of one excursion during this 10 day period, since, due to the presence of a tall locked chain-link fence surrounding the property, successive daily visits are not reasonably assumed at this site.

Similarly, for lead, exposure to surface soils containing 4000 mg/kg by the ingestion and inhalation routes would produce a total acute intake of 8.7 ug/kg/visit. In comparison, U.S. EPA (1980) established an acceptable daily intake of 1.4 ug/kg/day for chronic lead ingestion in drinking water. Therefore, acute lead exposures under worst case site conditions would pose minimal risk.

An additional avenue for human exposures is consumption of fish taken from the bay which connects with Crab Orchard Lake. PCB-contaminated sediments provide a source for bioaccumulation of residues. Associated risk levels due to consumption of fish from Crab Orchard Lake are presented in Section 38.4, Environmental Effects for Crab Orchard Lake.

Restricting access to the landfill also reduces the potential for deer to acquire site contaminant residues which could be ingested with venison. Although deer have been seen at the landfill, previous studies (see Section 2.7) have shown no contamination in deer hunted in the vicinity of this site.

Wildlife Risks

Estimates of total PCB intakes in receptor species of wildlife are given below:

Estimated Daily PCB Intake

Species	Body Weight kg	Inhalation (mg/kg/day)	Ingestion Food (mg/kg/day)	Ingestion Water (ng/kg/day)	Total Exposure mg/kg/day
Mallard	1.0	NA	30.4	1.9	30.4
Rabbit	1.0	0.026	45.6	1.9	45.6
Mouse	0.03	1.09	30.4	1.9	31.5

Note: See Table 24-1, Section 24.4.2.2 for assumptions.

NA = not applicable

These estimates indicate that wildlife exposures to site-related PCB residues may be greatest among small mammals on the landfill such as rabbit, mice, chipmunks, and the like. These animals will be exposed primarily via ingestion of contaminated soil and vegetation dust while burrowing, grooming, and feeding on dust-bearing seeds and invertebrates. Given the broad range of demonstrated possible PCB effects, the potential for interspecific sensitivity, and limited data on effects of PCBs on wildlife species, it is difficult to gauge the significance of these exposures. Using data from controlled tests with laboratory strains of animals, intake levels as high as 21000 ug/kg/day may not be sufficient to induce liver pathology, reproductive success, teratogenic effects, or neoplasia (U.S. EPA 1980). However, the

possibility for behavioural, immunological, or other subtle effects at these exposure levels which decrease competitiveness and survival in wild species cannot be discounted. Also, assuming that ingested PCBs are accumulated in these small mammals to some degree, a complete, non-quantifiable exposure pathway may exist for small predators such as skunks, weasels, and hawks which are able to gain access to the landfill while hunting.

Lead exposure in small burrowing mammals and other terrestrial wildlife may also present risks from chronic effects. For instance, using the previous exposure scenarios and a 4,000 mg/kg lead soil content, a chronic intake of 38 mg/kg/day is estimated for a mouse. These exposure levels could very likely produce behavioural, reproductive and other chronic effects if maintained at the assumed levels.

In contrast, exposure of aquatic wildlife or organisms feeding in the lake bay adjoining the landfill appear to be low. Ingestion rates of PCBs estimated to occur by consumption of contaminated fish by piscivorous birds and mammals are discussed in Section 38.4. Adverse effects to benthic organisms or species using PCB-contaminated sediments as a spawning substrate are possible but cannot be predicted from available data.

36.4.3 Analysis of Uncertainties

Among the areas of uncertainty in this analysis are the adequacy of the analytical data base and the set of assumptions that were selected to analyze human risk at the site. PCB analyses were sufficient to derive a representative soil level for the exposure assessment. However, lead content of site soil was extremely variable, with most sites showing only

background values. In order to conservatively estimate potential risks, a soil lead content of 4000 mg/kg was selected as representative, based on a range of samples with elevated lead of 205 to 8270 mg/kg. Other assumptions were also biased towards an upper bound worst case, in order to be most protective of public health. Some of these include consideration of trespassing across a chain link fence around the site, the presence of significant amounts of exposed contaminated soil, and ingestion of very large amounts of soil during these incursions. TCDF was not chosen as a site indicator contaminant for analysis due to lack of information on whether a toxic isomer of TCDF, such as the 2,3,7,8- isomer, was present.

Further, there is no evidence at present that TCDF poses a non-threshold carcinogenic risk comparable to PCBs, or that significant amounts of TCDFs bound to soil can be absorbed if ingested (Paustenbach, 1986). If this situation should change in the future, a revised risk assessment should be made.

Two other principal areas of uncertainty exist in addressing the risks posed to wildlife by chronic exposure to PCBs at the landfill site. A lack of documentation on the effects of PCBs on wildlife species which might be found on the site necessitated the use of studies involving laboratory rodents and rabbits as surrogates. The relative sensitivity of these species is unknown. Similarly, the chronic effects of benthic aquatic organisms to residue levels of PCBs found in the bay connecting with Crab Orchard Lake are largely unknown. Virtually all controlled toxicity assays are performed with toxicant in the water column only, which may be inadequate to determine the risks to benthic organisms exposed to contaminants in bottom sediments and interstitial waters.

Two main problems with conducting assays with non-water soluble contaminants are: 1)the water does not acquire a high enough concentration of the contaminant to indicate that there is a problem, usually measured by mortality rates; and 2)bioassays are short term, 96 hours or less, and severe chronic impacts cannot be measured in this short time span. Interstitial water spun off in a centrifuge is now being used in bioassays to determine the toxicity of non-water soluble contaminants.

36.5 Preliminary Remedial Alternatives

The sampling data and the risk assessment described in the previous sections indicate the presence of contaminants in soil which will require remediation. Contaminated areas containing PCB levels above 50 mg/kg are have been highlighted in Figure 37-4 and include some soil pockets up to 12 feet depth in the landfill. The sediments in the lowland area northeast of the landfill contained PCB levels below 5 mg/kg, with the exception of three samples (see Figure 36-3). The samples collected from the lake embayment adjacent to the site contained PCBs below 1 mg/kg with the exception of one sample (4.09 mg/Kg) taken close to the shoreline. Remediation of the site might address alternative measures for controlling potential leaching of contaminants to surface runoff and lake waters.

The criterion for cleanup for PCB contamination in surface or drinking waters, grazing lands, and vegetable gardens is set by EPA on a site specific basis and will be addressed in the FS. Based on the assumptions developed in Section 36.4, the contaminant levels do not pose long-term risks to chronically exposed wildlife or repetitive site visitors. It should be noted that these long-term, repetitive exposures are highly unlikely for humans, since the site

is somewhat isolated and site access is restricted by a tall fence and locked gates. However, if this scenario were possible, it is estimated that the risk levels to humans due to PCBs could be 40 to 400 fold greater than would be considered acceptable. Thus, reducing exposure to surface PCB levels in soil to between 7 to 70 mg/kg will reduce risks to an acceptable range of 10^{-6} to 10^{-5} for humans. This level is also 14 to 1.4 lower in magnitude than the no observed effect level for protection of the most sensitive species of wildlife from chronic effects (see Section 24.4.2.2). Exposures to wildlife under an example cleanup level of 50 mg/kg PCBs were presented in Section 24.5.

Other contaminants detected in the soils above the levels for Refuge background include lead and mercury; however, remediation of the site for PCBs will also include those areas contaminated with lead or mercury. The ground water samples contained traces of PCBs below 0.045 ug/L. Total chromium in one well sample exceeded the standard but the corresponding dissolved concentration was below the standard. Some alternatives for remediation of contaminated Lake water in the embayment area adjacent to the Landfill might also be addressed as part of the remedial efforts for this site. Lake samples offshore from the Area 9 embayment (see Section 38) contained PCBs in the range of 0.008 to 0.019 ug/L in water columns and up to 77 ug/kg in sediment samples; however, only one water sample exceeded the ambient water criteria for aquatic life protection of 0.014 ug/L (24-hour average). All water samples were below the AWQC acute toxicity level of 2.0 ug/L for protection of aquatic life.

A summary of the potential remedial response actions and associated technologies for this site is presented in Table 2 of the Executive Summary. Remedial measures including excavation, capping, regrading, revegetating, and

surface water diversion will be evaluated in the FS for this site. A brief discussion of the potential responses is included below.

Limited Site Access

Currently the site is enclosed by a tall fence which remains locked at all times. Access to the site is further limited by a locked gate at the entrance of the access road to the Landfill. Continued site access limitations will greatly reduce the potential for human and/or wildlife exposure via the water or direct contact with soils/sediments. The site may require site use limitation until all contaminated materials have been removed or adequately contained.

Surface Water Control

Dikes or trenches for diversion of surface runoff could be constructed in the lowland area and around the landfill to preclude contamination of lake waters.

Removal or Containment of Soil and Sediment

Contaminated soil might be excavated and removed for treatment on-site or off-site or regraded and contained on-site in a secure landfill.

Based on the sampling results from this RI, dependent on depth of contamination, soil to a depth of 6 or 12 ft may require removal or containment. Containment or removal of selected sediments will reduce the potential for surface and ground water contamination. Clean fill will be used for regrading and capping. Several abandoned munitions storage bunkers on the Refuge could be used as industrial cells if retrofitted for containment of treated wastes.

On-Site or Off-Site Treatment

Technologies such as incineration, vitrification, solidification or fixation should be considered for PCB contaminated soils.

Monitoring

The remedial response might include periodic sampling and analyses of soil/sediments and of five site monitoring wells for chromium (waters only), lead and PCBs. Follow-up studies might begin immediately after remediation to verify the adequacy of the cleanup.

36.6 Conclusions and Recommendations

It can be concluded that the Area 9 Landfill Site is impacted, with the primary pollutants being PCBs, lead, and mercury. Exposure to the site has been minimized by a chain-link fence. It is recommended that remedial alternatives for this site be evaluated in the FS. Potentially applicable remedial measures include removal or containment of contaminated soil and sediments, as well as ground water and surface water monitoring.

SECTION 37 - SITE 33, AREA 9 BUILDING COMPLEX

37.1 Site Description

Further information on Area 9 can be found in Section 36.1. Site 33 consists of the Area 9 Building Complex currently occupied by Olin Corporation and used primarily for the manufacture of explosives. (See Figure 37.1). From 1946 to 1962, Site 33 was occupied by Sangamo Weston, Inc., Capacitor Division to manufacture power factor capacitors, AC motor run capacitors, and a variety of DC capacitors.

The components utilized by Sangamo in its operations were of various types and included aluminum, electrolytes, mica, and silver and lead foil. The Division also manufactured small transformers that used mineral oil as a dielectric. Subsequently, Olin Corporation has been using the industrial facilities at the site to manufacture explosives.

Previous soil investigations at Area 9 include one study conducted by Envirodyne Engineers for the Olin Corporation in 1984. The focus of that study was to identify the extent of PCB contamination in soil. The analytical results (PCB analyses) are presented in Figure 37-1. The soil samples collected adjacent to Buildings I-1-23 and I-1-2 contained PCB levels above 50 mg/kg wet weight. Some isolated samples collected along the sides of the access road to the Area 9 Landfill also contained elevated PCB concentrations.

37.2 Site Investigations

37.2.1 Phase I Site Investigations:

The objective of the Phase I soil sampling was to define the horizontal and vertical limits of contamination. The sampling locations were selected based on data from previous investigations for Olin Corporation and

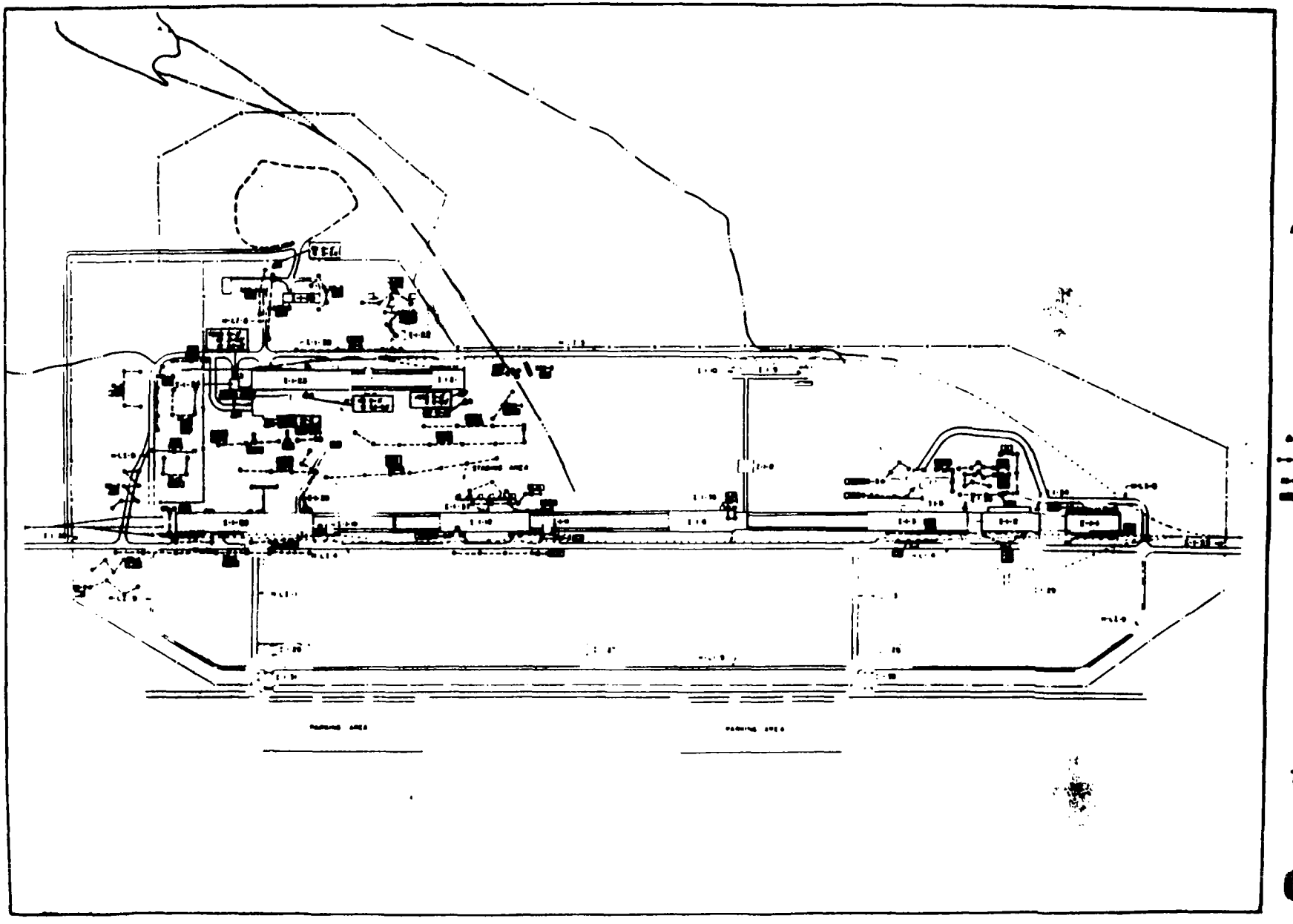


FIGURE 27-1

SITE 33

AREA 6 BUILDING COMPLEX
(OLIN DATA)

included, drainage pathways (located from aerial photographs), locations in close proximity to buildings, and transportation routes used to dispose of solid wastes. A total of 188 individual soil samples were collected, including surface, 0-1 ft., 1-2 ft., and 2-3 ft. depths. These samples represented 102 distinct locations within the Complex. (See Figure 37-2).

37.2.2 Phase II Site Investigations:

Additional soil samples were collected from 61 new locations and at greater depths (up to 6 ft.) for selected Phase I locations. Phase II was performed to more clearly define the extent of contamination. These samples were collected adjacent to contaminated areas identified in Phase I, at deeper locations, or from areas downgradient of Phase I samples, and also from several drainage ditches to trace the extent of migration. (See Figure 37-3). The soil samples were analyzed for PCBs; in addition, three of these soils were analyzed for the full list of priority pollutants. Three ground water monitoring wells were installed and sampled. These wells were installed to depths between 14 and 20 feet within silt and fine sand soils and were screened at the lower five feet of the boring. The ground water samples were analyzed for the full list of CLP organics, nitrosoamines, explosives, metals, and cyanide.

37.2.3 Site Hydrogeologic Characterization

37.2.3.1 Site Geology

Information obtained from the subsurface soil boring and wel installation program indicates that the site is immediately underlain by silty clay at least 25 feet thick. Since deeper well installations were not performed at this site, it is not possible to describe with any degree of

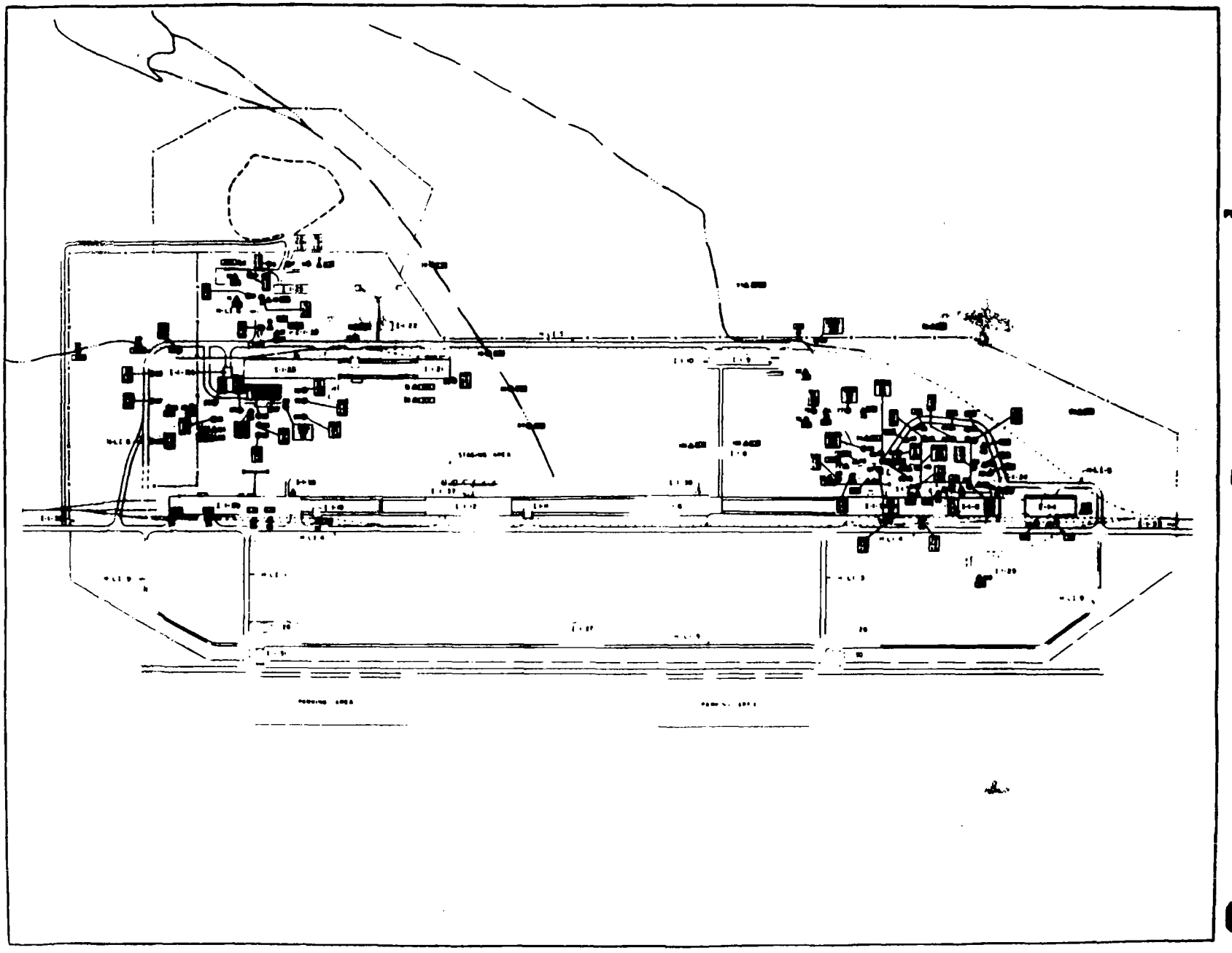


FIGURE 37-2

SITE 33

PHASE I SAMPLING LOCATI

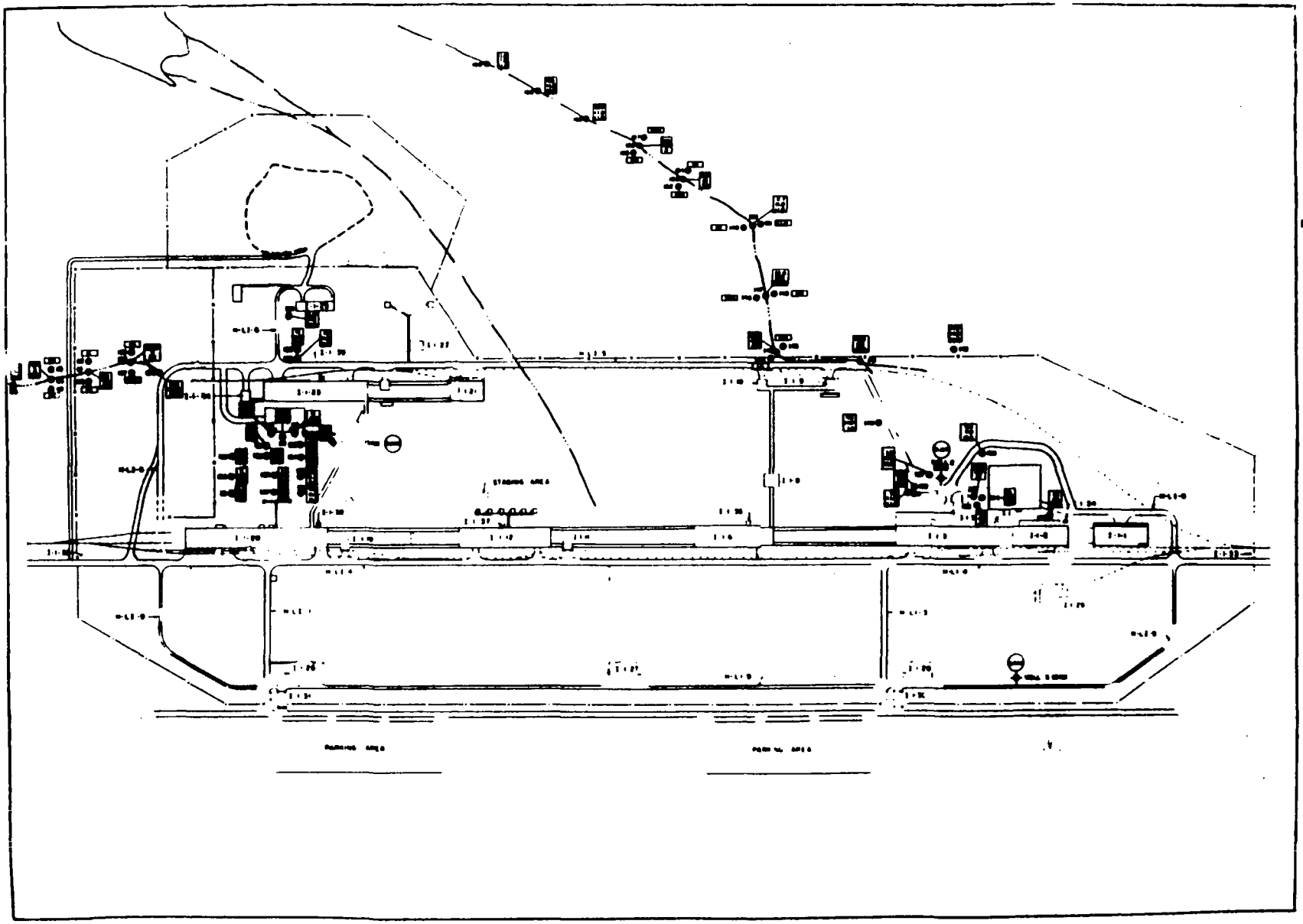


FIGURE 27-6

SITE 33

PHASE II SAMPLING LOCATIONS

certainty the soil types and thicknesses occurring beneath the silty clay. However, information from nearby Site 36 (Area 9 Landfill) indicates that the sand layers occurring there may continue beneath this site. Similarly, although bedrock was not encountered during drilling at this site, extrapolation of that surface between known data points on hydrogeologic cross-section A - A' (Figure 4-2) indicates bedrock may occur at a depth of 70 feet.

37.2.3.2 Site Hydrogeology

Occurrence of Ground Water

Ground water occurring within the three site wells which screen the shallow ground water table indicate an upper water table occurs about 6 to 14 feet below ground surface. Ground water elevation data collected in June 1986 indicates a higher water table of 5 to 8 feet occured during that period. Since no deeper wells were installed at this site, the nature of the ground water occurring in deeper portions of the unconsolidated sequence could not be evaluated.

Ground Water Flow Conditions

Ground water elevations obtained from the site wells on June 18-19, 1987 were contoured and are shown on Figure 36-4. Ground water flow directions interpreted from the figure indicate flow to occur both northeast and southwest off of a divide located near the southeastern end of the Olin Complex. The reason for the reversal of ground water flow to the southwest is most likely the presence of a shallow intermittent stream and swale located in this area. Surface water receives ground

water discharge from the southwestern portion of the site, then flows north into Crab Orchard Lake.

Ground water flow velocity in the northeast direction toward Crab Orchard Lake was calculated for this site. No velocity calculations could be made to the southwest due to limited data points. The hydraulic gradient (i) to the northeast averaged about 0.011 ft/ft. The average hydraulic conductivity (K) from wells 33-341 and 33-342 was calculated to be 1.3×10^{-5} ft/sec, or 1.12 ft/day. Porosity was assumed to be about 0.035 ft/day, or 12.8 ft/year, from data collected in June 1987.

37.3 Analytical Results (See Appendix I, Page 30)

37.3.1 Phase I Analytical Results:

Figure 37-2 shows the sampling locations and detected PCB levels in soils. The results supported the findings of Olin Corporation, in which the most significant locations for PCB contamination are adjacent to Buildings I-1-2 and I-1-23 where PCB concentrations exceeded 1,500 mg/kg. The results also identified two drainage ditches where contaminated storm runoff or spills have caused PCB migration from the site. Apparently, these drainage routes receive runoff water from areas surrounding Buildings I-1-2 and I-1-23.

A third drainage ditch originating in the Complex which passes east of the landfill and discharges to Crab Orchard Lake did not show evidence of contamination (PCB concentrations below 1 mg/kg). The old roadway access from the Building Complex to the landfill showed only isolated sampling locations where PCB concentrations exceeded 25 mg/kg. The locations where PCB levels exceeded 25 mg/kg were limited to the surface and/or upper 1 ft. of soil near the landfill access.

The soils collected along the west face of Building I-1-23 ranged from 500 to 120,000 mg/kg wet weight PCBs at the surface. Three soil cores collected within 30 ft. of the building contained elevated PCB levels to a depth of 3 ft. PCB concentrations in most samples collected from the lawn further from the building exceeded 50 mg/kg at the surface (0-1 ft.), but were below that criterion in the subsurface (1-3 ft. depth) samples. Two surface (0-1 ft depth) samples collected away from the building along the drainage ditch to the north contained PCB concentrations of 1200 and 1300 mg/kg wet weight.

The extent of PCB contamination (above 50 mg/kg wet weight) in the vicinity of Building I-1-2 was, for the most part, limited to the immediate areas surrounding the building and up to the access road. Some grab samples collected from either side of the access road and one collected close to a drainage route toward the lake also contained PCB concentrations that exceeded 50 mg/kg. Four soil cores collected at a depth of 3 ft. contained PCB concentrations in excess of 50 mg/kg also.

The results for PCDD and PCDF isomers in soils are presented on page 30A of Appendix I. A separate data listing is included to illustrate the comparison of actual dioxin/furan to PCB ratios compared to the ratios which would be expected based on the corresponding PCB concentration for that sample. The purpose of calculating this ratio is to determine if dioxins/dibenzofurans are elevated due to burning products of PCBs. If this were the case, then the dioxin/furan to PCB ratios would be higher than the expected normal ratios for PCBs. The Table lists a concentration for each peak detected in the scan (Isomer No.) and a total concentration for each compound. This total concentration was compared to the average PCB concentration detected in the same sample to develop a ratio of PCDD

or PCDF to PCBs. An average PCB concentration was used because the analyses were conducted in duplicate (one reported by ETC Laboratories and one reported by OBG Laboratories).

Based on studies conducted by T. Sawyer and S. Safe et.al.(1982, 1984, 1985) relative to PCB isomers and congeners, the expected fraction of PCDD and PCDF isomers associated with a measured PCB concentration can be calculated. The 'equivalent' fraction of dioxins and furans is determined by a conversion factor or Keq value. The Keq value is derived based on 1.00E-05 for 2,3,7,8-TCDD.

The results for Area 9 Building Complex showed positive detections above the ratios which would be expected for the sample collected in front of Building 1-1-23. The dibenzofuran isomers ranged from 28 to 249 ug/kg; the highest concentrations were associated with HxCDF (249 ug/kg), and PCDF (158 ug/kg), whereas the more toxic TCDF was detected at 28 ug/kg. The dioxins ranged from less than 0.11 for TCDD to 169 ug/kg for OCDD. The other soils contained traces of OCDD (6.2 to 9.7 ug/kg), and PCDF (less than 0.08 to 0.17 ug/kg) above the detection levels.

Four soil samples (0-1 ft. depth) screened for full priority pollutants did not contain any other organics at concentrations above the detection limits. However, the volatile and semi-volatile organic data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (See Exhibit B); some compounds which were not detected may in fact be present.

All measured concentrations for metals were similar to those detected at the control sites. The metals screening data are included in the data listing as estimated values.

37.3.2 Phase II Analytical Results:

Three ground water samples were collected at the Building Complex. PCBs were detected at concentrations of 0.093 ug/L (Well 33-340, adjacent to Building 1-1-23), 0.114 ug/L (Well 33-341, downgradient from Building 1-1-2), and 0.006 ug/L (Well 33-342, on the opposite end of the Complex, close to Building 1-1-28). These concentrations exceed the Ambient Water Quality Criteria for human health. In addition, wells 33-340 and 33-341 contained concentrations of chromium (113 ug/L and 50.0 ug/L respectively that exceeded the Illinois Public Water Supply Standards and Federal MCLs. However, the dissolved chromium levels below 1.3 and 1 ug/L for wells 33-340 and 33-341 respectively, were below the standards.

Traces of volatile organics were also detected in Well 33-341, including t-1,2-dichloroethene (404 ug/L) and trichloroethene (906 ug/L). The detected concentration of trichloroethene exceeded the AWQC of 18.4 ug/L for protection of human health. All other ground water parameters were below regulatory standards.

Three soils from the Phase II sampling were analyzed for full CLP organics and metals. Some organics (other than PCBs) were detected, including 1,2,4-trichlorobenzene (23,500 ug/kg in Sample 33-222), and 2-chloronaphthalene (6820 mg/kg in sample 33-270). Metal concentrations were generally within the ranges found at the control sites with the exception of one soil sample which contained 1400 ug/kg mercury (sample 33-291 at 0-1 ft. depth). Mercury was not elevated in the other soils analyzed for metals.

The detected PCB concentrations supported the findings from Phase I and provided the necessary data to better define the areas of contamination. (See wet weight PCB concentrations in Figure 37-3). The

results showed contamination along the two drainage paths from Buildings 1-1-23 and 1-1-2. Some samples contained levels above 50 mg/kg wet weight up to 3 feet depth, but PCB concentrations were well below that level in samples collected at further distances from the buildings. Core and surface sediment samples along the northeast drainage route downstream from Building 1-1-2 were taken up to where the creek enters Crab Orchard Lake. The last soil core (location 160) did not contain detectable PCBs to 1 mg/kg, from a high 4,100 mg/kg wet weight at the surface of the first soil core from the drainage path.

Soil cores to 3-ft. depths and surface samples were also taken from the drainage route from Building 1-1-23. The PCB levels along this drainage ditch ranged from 4,780 mg/kg to below 1 mg/kg wet weight. In addition, the surface sediments collected from the Area 9 embayment as part of Site 32 provided characterization for the area where the creeks entered the lake. Most sediments from the embayment contained PCB levels below 5 mg/kg wet weight, with the exception of two samples close to the lake boundary which contained 6.1 and 18 mg/kg PCBs. Figure 37-3 shows the sampling locations and the PCB concentrations along the ditches for the Building Complex. Soil cores collected from the lawn in front of Building 1-1-23 supported the Phase I findings and showed that PCB levels were elevated in the immediate vicinity of the building up to a depth of 6 ft.

The area around Building 1-1-23 is currently enclosed with a tall chain link fence. In addition, the immediate vicinity of the Building is roped off. The contaminated area is not mowed therefore the grass is tall and thick. PCB concentrations were well below 50 mg/kg in the soil samples (up to 3 foot depth) collected from the lawn area about 100 ft

from the building. Similarly, soil cores from the 3 ft. and 6 ft. depths adjacent to the concrete pad outside of Building 1-1-2 contained PCB levels that exceeded 50 mg/kg wet weight. The cores collected further from the building exhibited high PCB levels only at the 0-1 ft. depths. See wet weight PCB concentrations in Figures 37-3 and 37-4.

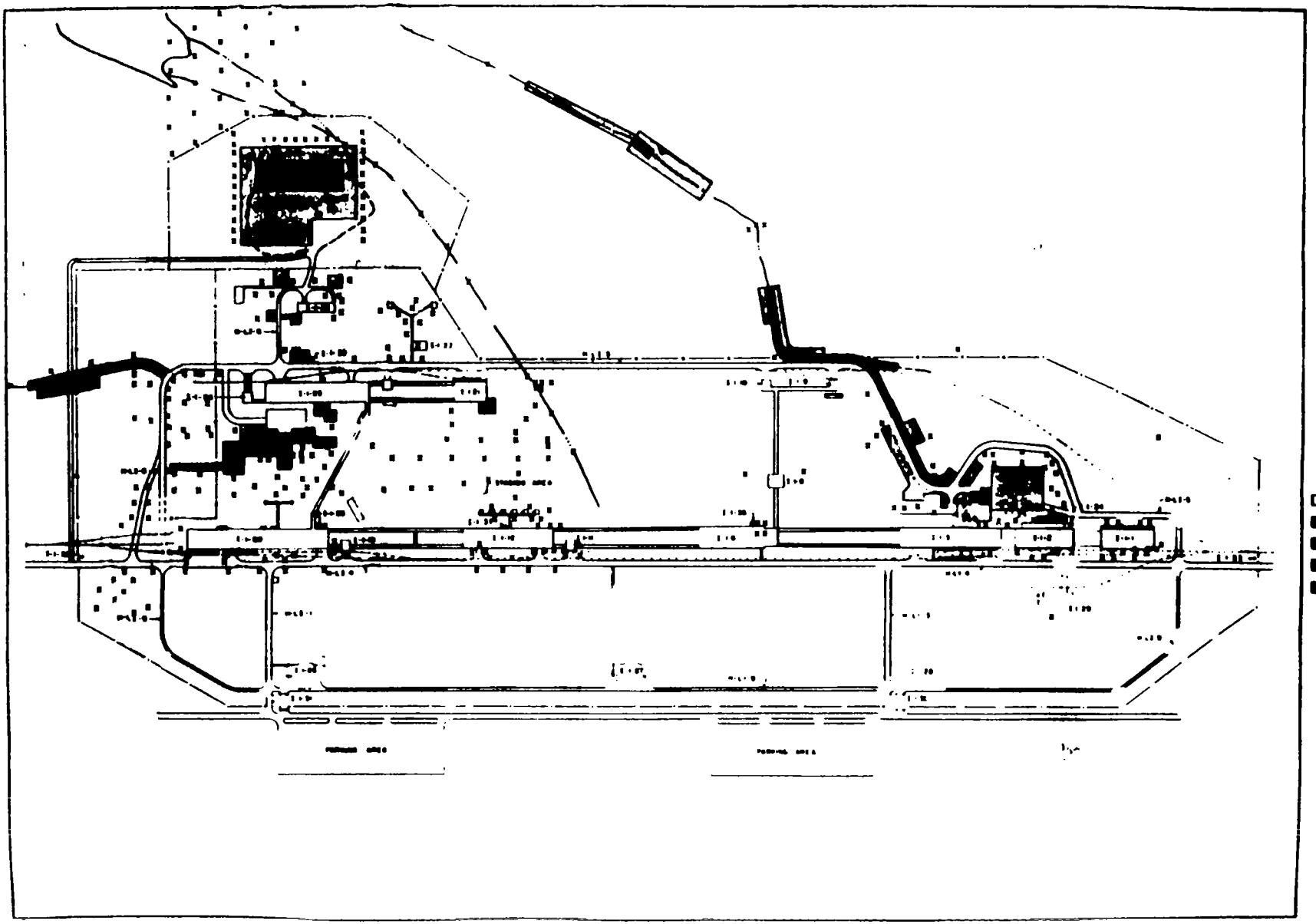
37.4 Environmental Effects

37.4.1 Qualitative Assessment

37.4.1.1 Source Evaluation

The results of the site investigations, as described in the preceding sections, determined that the Area 9 Building Complex has been used in the manufacture of capacitors, and explosives. Soil analyses of the site have shown the presence of several materials including PCBs, trichlorobenzene and chloronaphthalene. Three soil samples at the site contained tetrachlorodibenzofuran (TCDF), isomers unspecified, ranging from 0.14 ug/kg to 26.3 ug/kg. An unspecified isomer of tetrachlorodibenzodioxin (TCDD) was detected in one of these samples at 0.09 ug/kg. Trichloroethene, PCBs and t-1,2-dichloroethene were detected in site ground water at levels up to 906 ug/L, 0.114 ug/L, and 404 ug/L, respectively. No other contaminants in excess of Illinois General Use Water Supply Standards were found or any other water criteria or standards.

The areas contaminated with PCBs at concentrations above 50 mg/kg are identified in Figure 37-4. The total area has been estimated to be approximately 13 acres. The principal areas involved are immediately adjacent to Buildings 1-1-23 and 1-1-2, and two drainage paths leading from them (Figure 37-4).



SECTION 38 - SITE 34, CRAB ORCHARD LAKE

38.1 Site Description

Site 34 is comprised of Crab Orchard Lake (See Figure 38-1), which was formed in 1940 by construction of a spillway across Crab Orchard Creek. The lake has a surface area of 6,965 acres, a maximum depth of 30 feet, and 72,525 acre-feet of storage capacity. The retention time is 0.8 years (Kelly and Hite, 1981).

Water enters the lake through several creeks, including the Crab Orchard Creek at the eastern end of the lake. Water exits the lake through the spillway at the western end and through use of 280,000 gpd by the Refuge.

The eastern section of the lake has been bordered by manufacturing operations since the 1940s. Additional background on the site can be found in the introductory sections of this report. Previous investigations for water and sediments of Crab Orchard Lake are described in Section 2.6.

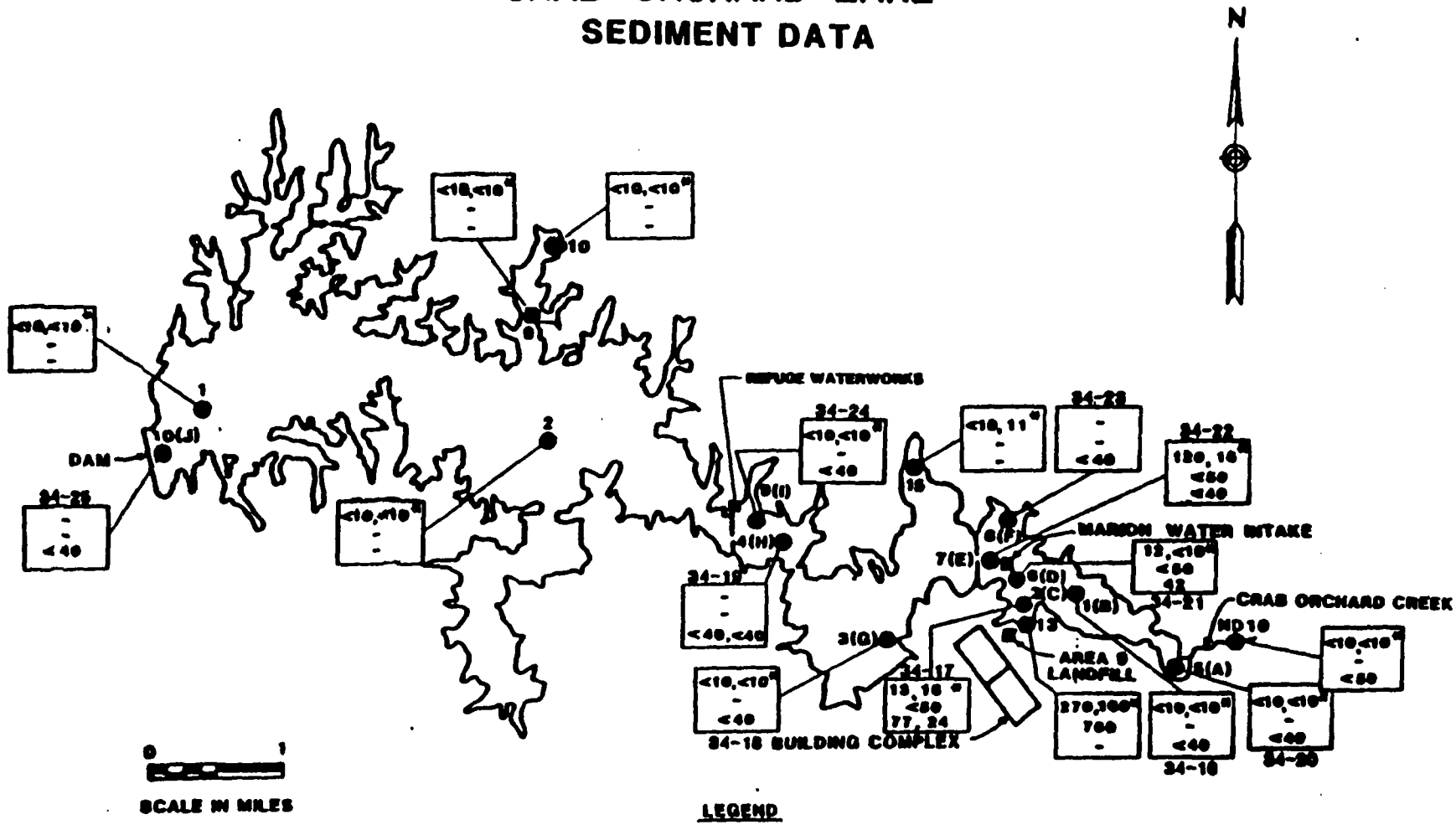
38.2 Site Investigations

38.2.1 Phase I Investigations:

Five water samples from current or potential drinking water sources were collected:

1. The Crab Orchard Refuge Treatment Plant Intake sample was collected at the intake structure under the grating and labeled as "Refuge Intake". The nearby Federal Penitentiary also derives its water supply from the Refuge water treatment plant.
2. The City of Marion WTP Intake sample at Marion Reservoir was taken by the dam at the intake structure and labeled as "Marion Intake".

CRAB ORCHARD LAKE
SEDIMENT DATA



LEGEND

● SAMPLE SITE

PCB CONCENTRATION, mg/kg (WET WEIGHT)

( )
( )
( )

MAY 26-28,1983 SITE, 1984)
MAY 21,1984 SITE, 1984)
DECEMBER 11-16,1986 – PHASE II SAMPLING

* REPLICATE SAMPLES

O'BRIEN&GERE ENGINEERS INC

3. The Marion Reservoir Auxiliary Intake sample at Crab Orchard Lake was taken from the lake prior to the intake structure; it is referred to as the "Marion Reservoir Intake."

4. The Refuge Finished Water sample was taken from the tap at the Refuge Fire Station and labeled "Refuge Treated" (also treated supply for Penitentiary).

5. The Marion Finished Water was sampled from the city water tap, and was labeled "Marion Treated".

These samples were analyzed for drinking water quality parameters and PCBs. The Marion Reservoir Auxiliary Intake has historically been used only once or twice within the last fifteen years, as stated in Section 2.5. According to the City Engineer, the City of Marion currently uses Herrin Lake as a backup supply, rather than Crab Orchard Lake. Crab Orchard Lake will no longer be used to supplement the Marion Reservoir except as a last alternative. It is thus not a current drinking water source, but a potential drinking water source.

Thirty fish composite samples were collected from Crab Orchard Lake including bass, carp, bullhead, and catfish species. Fish samples consisted of single species composites of the edible tissue portions of two to five fish. The fish sampling locations are shown on Figure 38-3. The analyses of the fish samples were performed as part of the Phase II investigation.

38.2.2 Phase II Site Investigations:

The five current or potential public water sources were resampled and analyzed for nitrosamines, PCBs, metals and cyanide. Ten composite water columns were collected from the lake at three depths (see locations in

Figure 38-2): at the surface, mid-depth, and near the bottom. These samples were analyzed for cadmium, chromium, lead, arsenic, cyanide, PCBs, and low-level nitrosamines. Ten grab sediment samples were collected and analyzed for semi-volatiles, pesticides, PCBs, metals, and cyanide. Thirty (30) fish composite samples collected in Phase I were analyzed during the Phase II investigations for pesticides, PCBs, lead, mercury and cadmium. Each composite consisted of two to five individual fish of a particular species, including composites of bass, bullhead, carp, and catfish. The analytical procedures were performed on a homogenized composite of the edible tissues for each sample point. Specific procedures for filleting (skin-on, skin-off), blending, and digesting were detailed in Addendum No. 3 to the QAPP, March, 1987.

38.3 Analytical Results (See Appendix I, Page 32)

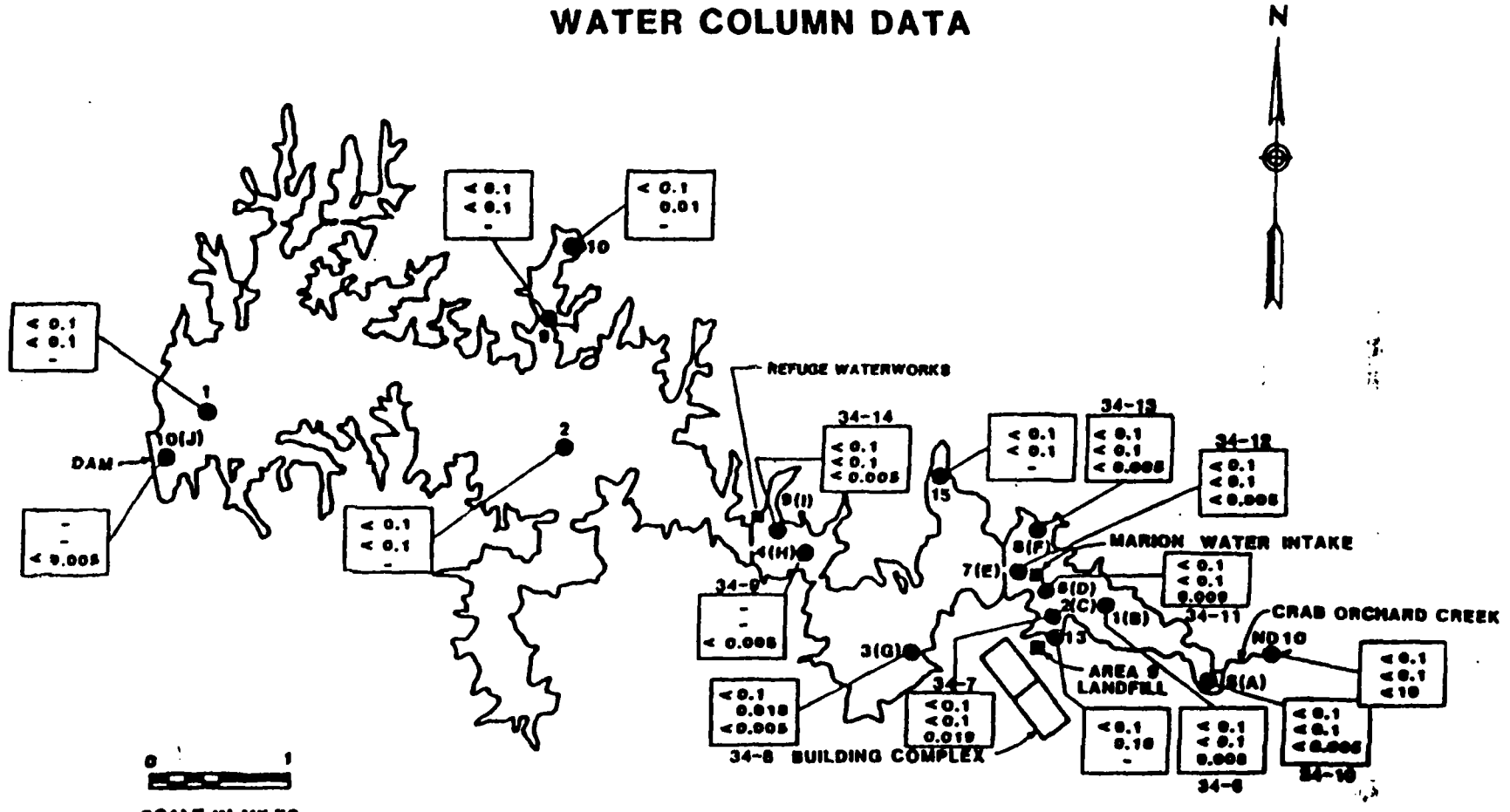
38.3.1 Phase I Analytical Results:

All the concentrations of detected contaminants in the current or potential water supply samples were below the Illinois Public Water Supply Standards and Federal drinking water standards with the exception of two samples which exceeded the Federal MCL of 0.05 mg/L for manganese. It should be noted that the water standard for manganese is established based on aesthetic concerns of taste, or odor; thus, exceedance of this standard does not constitute a health concern. These samples represented the Marion intake (0.09 mg/L) and the Refuge treated water (0.28 mg/L). As shown in Exhibit B, the Phase I metals analyses are questionable because of QA/QC deficiencies. The City Marion treated water contained concentrations of bromodichloromethane (17 ug/L) and chloroform (180 ug/L) which exceeded the Federal MCL of 100 ug/L for total

trihalomethanes for the Refuge Treated Water Supply slightly exceeded the Federal Drinking Water MCL of 100 ug/L. These compounds are likely formed as a result of chlorination. Corrective measures have been taken at the Refuge Water Treatment Plant to reduce levels of trihalomethanes. Acetone, methylene chloride and isophorone were also detected but were also present in the method blanks. Cyanide levels (0.09 mg/L but not detected in the duplicate) in the sample from the City of Marion treated supply were above the Illinois General Use and Public Water Supply standards, but below the lifetime health advisory of 154 ug/L for cyanide in drinking water.

Three lake waters contained low parts per trillion levels of PCBs. Locations 1B, 2C, and 6D (Samples 34-6, 7, 11) contained 0.008, 0.019 and 0.009 ug/L respectively; but only sample 34-7 at location 2C exceeded the chronic AWQC of 0.014 ug/L for protection of aquatic life. PCBs were not detected in any of the other lake water samples (detection limit 0.005 ug/L). (See Figure 38-2.) Traces of arsenic (2.0 to 3.4 ug/L), chromium (1 to 7 ug/L) and lead (1.2 to 9.2 ug/L) were detected, but only arsenic exceeded the human health AWQC of 0.0022 ug/L. The cyanide concentration (0.29 mg/L) exceeded the Illinois standards and the AWQC criterion in one water sample (Location 1B or No. 34-6); cyanide was undetected in all other samples at a detection level of 0.05 mg/L. Likewise, in one water sample (Location 10J, No. 34-15), the mercury concentration (0.4 ug/L) exceeded the ambient water quality standard for human health (0.144 ug/L). Sediment analyses indicated the presence of trace quantities of base/neutral/acid compounds. Di-n-butyl phthalate was detected in all ten sediments, ranging from 1,000 - 2,240 ug/kg wet weight; however, this compound was also detected in the QA/QC blanks.

CRAB ORCHARD LAKE
WATER COLUMN DATA



O'BRIEN & GERE ENGINEERS INC

trihalomethanes. The formation of these compounds can be attributed to chlorination from the disinfection of drinking water supplies and is not indicative of off-site contamination sources. The City of Marion has instituted modifications to the treatment process and is now in full compliance for trihalomethanes, according to Illinois EPA's drinking water group. However, the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data. (See Exhibit B). The positive detections reported are thus estimated values and some compounds which were not detected may in fact be present.

38.3.2 Phase II Analytical Results:

The water samples collected from Crab Orchard Lake and from various raw and treated water supplies or potential water supplies were compared to the Illinois and Federal drinking water standards. Raw water supplies were also compared to the Ambient Water Quality criteria.

Two raw water supplies (Refuge Intake and Marion Reservoir Auxiliary Intake) contained trace amounts of methylene chloride (24 and 6 ug/L), barium (36 and 33 ug/L), and lead (4.8 and 8.5 ug/L). The detected concentrations for barium and lead are typical of raw surface water supplies, and are within the relevant standards. PCB concentrations were below the detection level of 0.005 ug/L in all of the current or potential drinking water samples. Methylene chloride, though detected, was also detected in the blank samples and may be a result of sample or laboratory handling. The treated water samples (Refuge treated water and City of Marion treated water) contained bromodichloromethane (11 and 4 ug/L respectively), and chloroform (94 and 35 ug/L). Total

Bis (2-ethylhexyl) phthalate was detected at 750 ug/kg in sample location #8F, No. 34-23. The lake control sample collected from the west end by the spillway dam (location #10J, No. 34-25), remote from any industrial areas of the Refuge, also contained phthalates (480 ug/kg wet weight), pyrene (450 ug/kg wet weight), and fluoranthene (510 ug/kg wet weight).

PCB concentrations were below the 40 ug/kg wet weight detection limit in all but two sediment samples (See Figure 38-1). Sample location #2C (No. 34-17) collected near the Area 9 Landfill area contained a PCB concentration of 248 ug/kg (291 ug/kg duplicate); while sample location #1B (No. 34-16) contained 104 ug/kg PCBs. Arsenic was detected in all ten sediments at concentrations ranging from 9.7 to 24 mg/kg, but duplicates and recoveries were outside control limits for these analyses. All other compounds analyzed were below the detection limits. The field data for the fish samples is presented as Table 38-1 and includes the weight and size for each fish sample collected. The analytical results for pesticides/PCBs, cadmium, mercury, and lead are presented in Table 38-2. Figure 38-3 shows the sampling locations and PCB results. Most parameters were below the detection levels, but concentrations of Aroclor 1254 ranged from below 0.4 to 6.4 mg/kg wet weight. Two carp samples, 34-29 and 34-30 from location #2C, contained 6.4 and 3.0 mg/kg PCBs respectively (3.9 and 4.4 mg/kg when re-analyzed); these concentrations are above the FDA action level of 2.0 mg/kg for PCBs. It should be noted that FWS conducted a split sample analysis for sample 34-29 and obtained a result of 1.2 mg/kg PCB wet weight, which is below the FDA action level, and is significantly below the earlier results for this sample. Fish sample 34-27 from location #1B contained 1.05 mg/kg wet weight of mercury. This concentration is only slightly above the FDA action level of 1.0 mg/kg.

TABLE 38-1

FIELD DATA FOR FISH COMPOSITE SAMPLES

SAMP No.	I.D.	LAB No	DUP/SPKE FWS	SPECIES	WEIGHT LBS.	LENGTH INCHES	WEIGHT LBS.	LENGTH INCHES	WEIGHT LBS.	LENGTH INCHES	WEIGHT LBS.	LENGTH INCHES	WEIGHT LBS.	LENGTH INCHES
438	34- 26	19170		CARP	3	18- 3/8	3	18- 3/4	4- 1/2	21- 1/2	4- 1/8	21- 1/8	2	16
439	34- 27	19171		BASS	1- 1/2	13- 7/8	1- 1/4	13- 1/2	2- 3/8	16	3- 7/16	18	1- 3/16	13
440	34- 28	19172	DUP	BASS	2- 1/4	15- 1/2	3- 9/16	17- 1/2	2- 1/2	16	1- 7/16	13- 7/8	1- 1/8	12- 3/4
460	34- 48	19192		BULLHEAD	0- 1/4	8	0- 1/4	7- 7/8	0- 9/16	10- 1/4	0- 5/8	10- 3/8	0- 1/4	8
460	34-		FWS	BULLHEAD	0- [illegible]8	9	0- 5/16	8- 7/8	0- 1/4	8- 1/4	6	9- 3/8	0- 1/4	8
461	34- 49	19193	DUP	BULLHEAD	0- 1/4	8- 1/2	7	9- 1/8	2	6- 5/8	0- 1/4	8	0- 1/4	6- 7/8
462	34- 50	19194		CATFISH	9	12	4- 3/4	22- 7/8						
441	34- 29	19173		CARP	2- 1/8	17- 1/4	3- 1/4	19	1-13/16	16- 1/8	2-13/16	18- 3/4	2- 1/2	18- 2/8
441	34-		FWS	CARP	1- 3/4	15- 1/8	1- 1/4	13- 7/8	1- 1/8	13- 3/4	1- 1/8	13- 3/8	1- 1/4	14
442	34- 30	19174	DUP	CARP	2- 5/8	18- 1/8	2- 3/4	18- 3/8	3- 3/4	19	2- 1/4	17- 1/2	1- 1/8	14- 3/8
443	34- 31	19175		BASS	1-13/16	15- 1/8	2- 7/16	17	1-13/16	14- 7/8	1- 1/2	13- 7/8	1	12- 3/8
463	34- 51	19195		BULLHEAD	0- 5/8	10- 3/4	0- 3/8	8- 3/4	0- 3/8	9- 1/4	0- 3/8	9- 1/8	0- 3/8	8- 7/8
464	34- 52	19196	DUP	BULLHEAD	0- 3/16	6- 1/2	0- 3/8	9	0- 1/4	8	0- 1/2	10	0- 3/8	9- 1/8
465	34- 53	19197		CATFISH	9	28- 7/8	8- 1/8	26	0- 7/16	31- 7/8	3- 9/16	20- 7/8	4- 3/4	23- 1/8
444	34- 32	19176		CARP	2- 1/4	16- 7/8	1- 3/16	14- 3/8	1- 5/8	15- 1/2	0-13/16	12- 1/8	1-13/16	15- 3/8
445	34- 33	19177	DUP	CARP	1- 7/16	14- 5/8	3- 1/8	19	1- 5/8	14- 5/8	1- 9/16	15- 1/4	1- 9/16	14
446	34- 34	19178		BASS	2- 7/8	17- 1/8	2- 1/4	16- 1/8	0- 5/16	11- 1/8	1	12- 3/4	0- 9/16	10- 1/4
	34-		FWS											
466	34- 54	19198		BULLHEAD	0- 5/16	8- 1/2	0- 5/16	8- 7/8	0- 1/4	8- 1/2	0- 1/2	9- 3/4	0- 1/2	10
467	34- 55	19199	DUP	BULLHEAD	0- 9/16	10- 1/8	0- 9/16	10- 3/8	0- 1/4	8- 1/8	0- 1/2	9- 1/2	0- 1/2	9- 7/8
447	34- 35	19179		CARP	3- 7/8	20- 1/8	1- 3/8	13- 3/8	2- 1/8	16- 1/4	2- 7/8	19	2- 3/8	16- 1/8
	34-		FWS	CARP	2- 1/8	15- 1/4	2	16- 1/8	2- 3/8	16- 3/4	1- 3/4	15- 1/4	2	15- 1/8
448	34- 36	19180		BASS	0- 3/4	11- 1/8	4- 7/8	20- 1/2	3- 1/2	18- 5/8	1- 3/16	13- 1/8	0- 7/8	11- 1/4
449	34- 37	19181	DUP	BASS	0-15/16	11- 5/8	1- 3/4	14- 1/2	1-13/16	14- 7/8	2	15- 1/2	2- 3/4	16- 7/8
472	34- 59	19200		BULLHEAD	0- 7/8	11- 1/2	0- 5/8	10- 1/2	0- 7/16	9- 1/4	0- 3/4	11- 1/4	0- 7/16	9- 1/4
473	34- 60	19201	DUP	BULLHEAD	0- 5/8	10- 5/8	0- 1/2	9- 3/8	0-13/16	11- 3/8	0- 3/8	8- 1/2	0- 1/2	9- 1/2
471	34- 61	19202		CATFISH	1- 3/4	13- 7/8	7- 1/2	26- 1/2	2- 3/4	20	5- 1/4	23- 1/2		
450	34- 38	19182		CARP	2- 7/8	17- 5/8	6	23- 1/4	2- 3/4	17- 5/8	5- 3/8	22- 3/4	2- 1/8	16- 1/2
451	34- 39	19183	DUP	CARP	2	6- 1/8	1- 7/8	16- 1/8	2-11/16	18- 1/8	2- 1/4	17	2- 1/2	17- 1/8
452	34- 40	19184		BASS	0- 3/4	11	0- 3/16	6- 7/8	1- 1/2	14				
452	34-		FWS	BASS	5- 1/16	20- 7/8	0- 5/8	10- 1/2	0-11/16	11	0- 3/4	11- 1/2		
453	34- 41	19185	DUP	BASS	1- 3/4	14- 7/8	1- 1/8	12- 1/2	3- 3/16	17- 1/2	2- 5/16	15- 1/2	1- 1/4	13- 5/8
468	34- 56	19203		BULLHEAD	0- 1/4	7- 3/4	0- 7/16	9	0- 1/4	8	0- 1/4	8- 1/8	0- 1/4	7- 3/8
469	34- 57	19204	DUP	BULLHEAD	0- 1/4	8	0- 5/16	9- 1/4	0- 1/4	7- 3/8	0- 1/2	10	0- 7/16	9- 7/8
470	34- 58	19205		CATFISH	2- 5/8	19- 1/8	10- 3/4	29- 3/4	4	22- 1/4				

TABLE 38-2 (p. 1 of 2)
Analytical Results for Fish (Edible Tissue)

Location:	LAKE 1B	LAKE 1B	LAKE 1B	LAKE 1B	LAKE 1B	LAKE 1B	LAKE 2C	LAKE 2C	LAKE 2C	LAKE 2C	LAKE 2C	LAKE 2C	LAKE 3B	LAKE 3B	LAKE 3B
Sample ID:	34-26	34-27	34-28	34-48	34-49	34-50	34-29*	34-30	34-31	34-51	34-52	34-53	34-32	34-33	34-34
Sample Type:	Carp	Bass	Bass	Bullhead	Bullhead	Catfish	Carp	Carp	Bass	Bullhead	Bullhead	Catfish	Carp	Carp	Carp
Sample Number:	438	439	440	460	461	462	441	442	443	463	464	465	444	445	446
PESTICIDES/PCB (ug/Kg wet wt)															
115 4,4'-DDD	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
116 4,4'-DDE	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
117 4,4'-DDT	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
118 Aldrin	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
119 Alpha-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
120 Arochlor-1016	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
121 Arochlor-1221	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
122 Arochlor-1232	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
123 Arochlor-1242	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
124 Arochlor-1248	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
125 Arochlor-1254	1100	< 400	480	420	< 400	1300	6400	3000	540	940	850	1700	840	990	450
126 Arochlor-1260	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400
127 Beta-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
128 Chlordane	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
129 Delta-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
130 Dieldrin	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
131 Endosulfan I	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
132 Endosulfan II	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
133 Endosulfan Sulfate	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
134 Endrin	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
136 Endrin Ketone	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
137 Gamma-BHC (Lindane)	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
138 Heptachlor	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
139 Heptachlor epoxide	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
140 Methoxychlor	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
141 Toxaphene	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400
METALS (mg/Kg wet wt)															
166 Cadmium	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
178 Lead	< 0.05	< 0.05	< 0.05	< 0.05	0.13	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05
184 Mercury	0.172	1.05	0.420	0.072	0.070	0.120	0.152	0.239	0.303	0.102	0.078	0.120	0.122	0.114	0.201

FDA ACTION LEVELS
PCB = 2 mg/Kg
Mercury = 1 mg/Kg

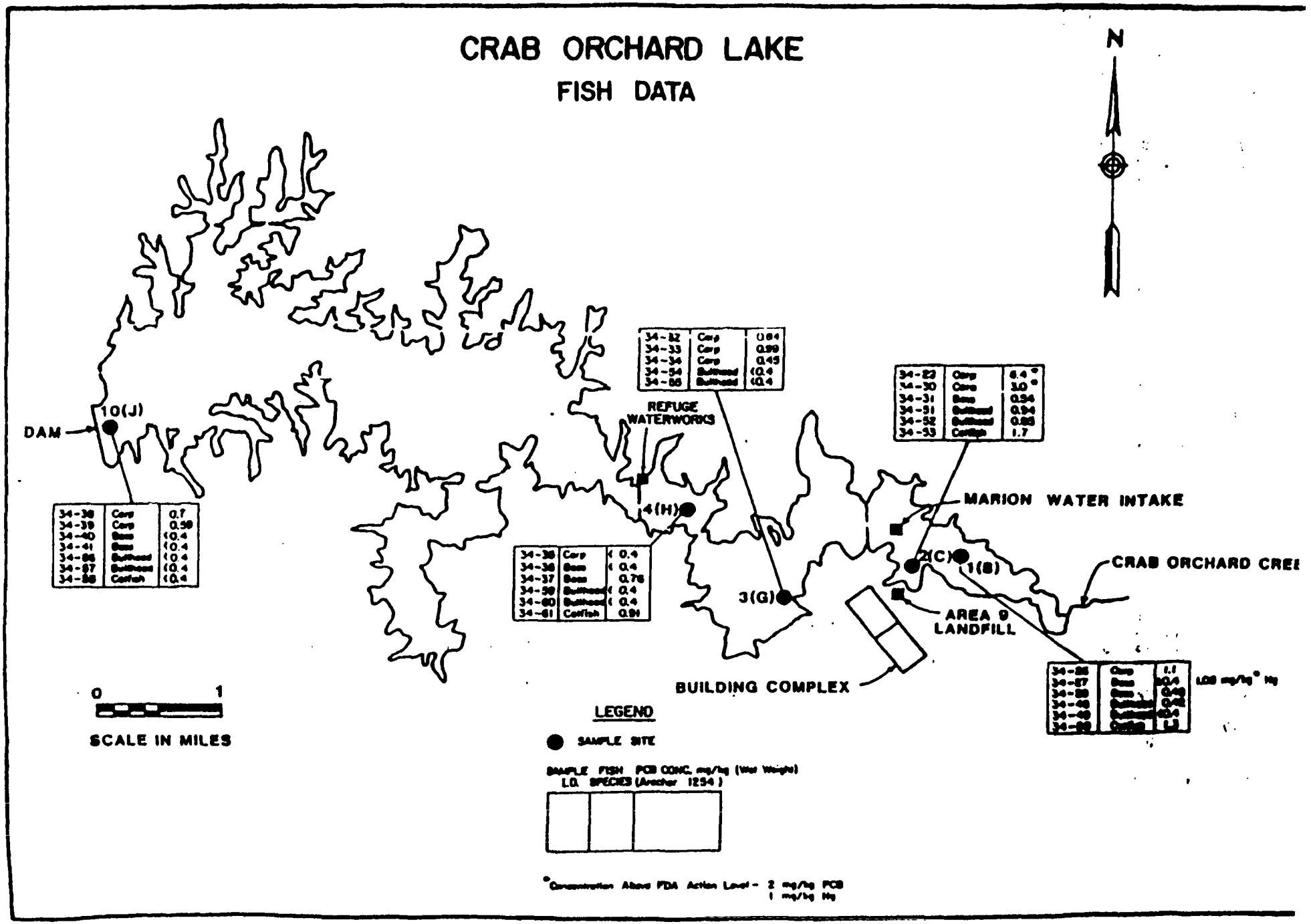
(*) FWS Split Sample No. 441 contained 1.2 mg/kg wet wt. PCBs.

TABLE 38-2 (p. 2 of 2)
Analytical Results for Fish (Edible Tissue)

Location:	LAKE 3G	LAKE 3G	LAKE 4H	LAKE 4H	LAKE 4H	LAKE 4H	LAKE 4H	LAKE 4H	LAKE 10J	LAKE 10J	LAKE 10J	LAKE 10J	LAKE 10J	LAKE 10J	LAKE 10J
Sample ID:	34-54	34-55	34-35	34-36	34-37	34-59	34-60	34-61	34-38	34-39	34-40	34-41	34-56	34-57	34-58
Sample Type:	Bullhead	Bullhead	Carp	Bass	Bass	Bullhead	Bullhead	Catfish	Carp	Carp	Bass	Bass	Bullhead	Bullhead	Catfish
Sample Number:	466	467	447	448	449	472	473	471	450	451	452	453	468	469	470
PESTICIDES/PCB (ug/Kg we															
115 4,4'-DDD	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
116 4,4'-DDE	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	1000
117 4,4'-DDT	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
118 Aldrin	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
119 Alpha-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
120 Arochlor-1016	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
121 Arochlor-1221	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
122 Arochlor-1232	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
123 Arochlor-1242	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
124 Arochlor-1248	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
125 Arochlor-1254	< 400	< 400	< 400	< 400	760	< 400	< 400	910	700	590	< 400	< 400	< 400	< 400	< 400
126 Arochlor-1260	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400
127 Beta-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
128 Chlordane	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
129 Delta-BHC	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
130 Dieldrin	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
131 Endosulfan I	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
132 Endosulfan II	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
133 Endosulfan Sulfate	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
134 Endrin	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
136 Endrin Ketone	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40	< 40
137 Gamma-BHC (Lindane)	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
138 Heptachlor	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
139 Heptachlor epoxide	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20
140 Methoxychlor	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200	< 200
141 Toxaphene	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400	< 400
METALS (mg/Kg wet wt)															
166 Cadmium	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
178 Lead	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	0.18	< 0.05	< 0.05
184 Mercury	0.078	0.052	0.116	0.094	0.217	0.055	0.061	0.055	0.091	0.079	0.057	0.108	0.034	0.063	0.168

FDA ACTION LEVELS
PCB = 2 mg/Kg
Mercury = 1 mg/Kg

(*) FWS Split Sample


CRAB ORCHARD LAKE
FISH DATA
N
DAM
10(J)
REFUGE WATERWORKS
4(H)
3(G)
2(C)
1(B)
MARION WATER INTAKE
CRAB ORCHARD CREE
AREA 9 LANDFILL
BUILDING COMPLEX
0
1
SCALE IN MILES
LEGEND
SAMPLE SITE
SAMPLE FISH PCB CONC. mg/kg (Wet Weight)
I.D. SPECIES (Aroclor 1254)
*Concentration Above FDA Action Level - 2 mg/kg PCB
1 mg/kg Hg

O'BRIEN & GERE

Mercury concentrations in the remaining samples ranged from 0.034 to 0.420 mg/kg. Positive lead detections were found in two bullhead samples, 0.13 mg/kg wet weight in sample 34-49 from location #1B, and 0.18 mg/kg wet weight in sample 34-56 from the control location #10J. Cadmium was not detected at a detection limit of 0.10 mg/kg wet weight in any of the samples. There are currently no FDA action levels for cadmium or lead.

Kohler and Heidinger (undated) of Southern Illinois University (SIU) have recently reported preliminary results of a survey for PCB residues in Crab Orchard Lake fish. The SIU study was conducted independently of other residue surveys including the RI. The investigators collected largemouth bass, channel catfish, carp, bluegill, white crappie, and gizzard shad (a non-food fish) from three areas of the lake, including the embayment adjacent to the Area 9 Landfill (identified as Site 10 in the report). In an effort to discern age-related effects on accumulation of PCBs in these species, young, intermediate, and old individuals of each species (selected on the basis of life expectancy for each species) were analyzed. The results of this survey are presented in Section 2.7, and are summarized briefly below.

With the exception of channel catfish and possibly carp, no clear correlation of residue PCB level in fillet tissue with fish age was detectable in the study by SIU. This could be due to the limited number of samples in each data set (three fish per group) and the lower lipid content of species other than catfish, carp, and possibly shad (a major determinant in PCB accumulation with both species and fish age). These data are similar to the data developed during this RI in that they show that fish in the eastern portion of the lake have higher PCB concentrations than those

taken elsewhere in the lake. The results also demonstrate that carp and channel catfish contain higher residues than other species analyzed.

PCB analyses in the SIU study appear to have been conducted at a lower analytical detection limit, possibly 0.1 mg/kg, because a number of the pooled data presented are lower than the 0.4 mg/kg detection limit used in this RI. The composite data for most of the species and collection locations show residue levels in the range of 0.2 mg/kg PCBs. As discussed later, this provides a basis for treating RI fish data of less than the detection limit of 0.4 mg/kg as 0.2 mg/kg, one half the detection limit, when calculating averages for the quantitative assessment.

Due to the preliminary nature of the data presented in the SIU report, they are not used in the quantitative assessment of PCB-associated risks. However, the data are in general qualitative and quantitative agreement with the data collected during the RI and other investigations.

As discussed in the review of previous investigations in Section 2.7, Stuart (1984) collected Crab Orchard Lake fish and bottom sediments from the Area 9 embayment for a survey of polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). This survey did not detect the highly toxic 2,3,7,8-TCDD isomer in either fish or sediments. The 2,3,7,8-TCDF isomer was found in fish at an average level of 15.2 ppt (mean of 10 fish samples) with a maximum level observed of 41 ppt. In sediment, TCDF was detected at a mean of 19 ppt (for 5 samples) and a maximum of 50 ppt. Other isomers of PCDD were reported at ppt levels, but the isomeric distribution was not identified.

38.4 Environmental Effects

38.4.1 Qualitative Assessment

38.4.1.1 Source Evaluation

The western portion of Crab Orchard Lake is a focal point for recreational activities in the Refuge and supports an extensive ecological community as well. The lake has been used previously as an auxiliary drinking water source for the City of Marion. As discussed in Section 2.5, this auxiliary intake has been used one or two times within the last fifteen years, and is presently non-operational.

As a result of a variety of manufacturing activities along the eastern portion properties for four decades, the eastern part of the lake has served as a potential recipient of industrial discharges, landfill leachates, municipal wastewater and other possible pollutant sources. The results of chemical analyses of lake water, sediments, and biota, as described in the preceding section and in Section 2.7, suggest that the eastern part of the lake has in fact received industrial discharges.

For the purpose of this risk assessment, potential human health effects will focus on ingestion of finished drinking water drawn from Crab Orchard Lake and of fish taken from the lake by recreational angling. Because of the wide variety of subchronic and chronic effects documented in laboratory tests, often at low exposure levels, PCBs were chosen as an indicator contaminant for the risk assessment. The toxicological and physicochemical properties of PCBs are summarized in Exhibit A and in the quantitative assessment portion of this report.

Dibenzodioxin and furan residues in fish and sediments were analyzed in a survey conducted in 1984 (Stuart, et al., 1984) as

presented in Section 2.7. The risks associated with the detection of PCDD/PCDF residues in fish are evaluated in Section 38.4.2.2, Quantitative Risk Assessment.

38.4.1.2 Transport Route Evaluation

a) Air: Due to the nature of these compounds, PCBs do not volatilize from water to a significant degree at the low concentrations found in Crab Orchard Lake. The air route is therefore considered non-functional.

b) Direct Contact: PCB residues have been detected in lake sediments. The direct contact exposure route is thus functional but will be considered in this analysis in the assessment of ingestion exposures.

c) Surface Water: PCB residues have been detected in some water samples from the lake, but not in drinking water prepared from it or in the raw water intakes to the treatment plant. Thus, the surface water route is functional for both humans and wildlife. This exposure path will be considered as a form of ingestion exposure in the following assessment.

d) Ingestion: PCB residues have been detected in aquatic biota taken from the lake (see also Previous Studies in Section 2.7). Therefore, the ingestion exposure pathway is complete for both humans and wildlife.

38.4.1.3 Receptor Evaluation

Human

Use of Crab Orchard Lake as a source of drinking water is very limited, serving only the Refuge and the Marion penitentiary at

present. Potential receptors therefore include visitors and employees of the Refuge as well as residents of the penitentiary. Analyses of Refuge finished drinking water failed to reveal the presence of PCBs. Therefore, the only potential human receptors for PCBs in the lake are consumers of fish captured by recreational angling. A hypothetical case is also evaluated where the backup supply for the City of Marion might be used to replenish the Marion Reservoir in an emergency.

Wildlife

Exposures of fish and wildlife to PCBs are complete for the direct contact/exposure route. Benthic and bottom-feeding organisms in the lake will receive the highest exposure. Exposure to PCBs is also possible for wildlife feeding on contaminant residues.

38.4.2 Quantitative Risk Assessment

38.4.2.1 Estimates of Surface Water Exposures

Previous surveys of Crab Orchard Lake for surface water residues of PCBs and other contaminants were presented in Section 2.7.2. With the exception of an elevated level of PCBs in a bay adjacent to the Area 9 Landfill (0.16 ug/L, Hite, 1984 and IDPH, 1976-1987) all water samples from previous surveys were below the detection limit of 0.1 ug/L. The Phase II investigation found only 0.009 to 0.019 ug/L PCBs in the Area 9 bay water and no PCBs were detected elsewhere at a detection limit limit of 0.005 ug/L PCB.

It is conceivable that Crab Orchard Lake could be used in the future as an emergency backup supply of water for the City of Marion to supplement the Marion Reservoir. This has occurred as

recently as 1981, but the alternate water source has since been switched to Herrin Lake. During the last withdrawal from Crab Orchard Lake in 1981, about 6 percent of the capacity of Marion Reservoir was replenished. To determine the health risks associated with such a use of Crab Orchard Lake, it will be estimated that such an event occurs every ten years, and that the replacement water contains 9 ng/l (ppt) of PCBs, based on a sampling of lake water closest to the auxiliary intake.

The resultant PCB concentration from replenishing 6 percent of Marion reservoir's capacity would be 0.5 ng/l, a level which is 10 times lower than the detection limit for PCB analyses in water. If such a diluted concentration were maintained by periodic replenishment from Crab Orchard Lake, a chronic surface water ingestion scenario would exist. The main concern for health effects from chronic exposure to PCBs is potential carcinogenicity based on controlled laboratory studies. In order to assess risks associated with such exposure, daily PCB exposure rates are estimated and integrated with a measure of the potency of PCBs to produce human cancer as extrapolated from animal response data.

Assuming a standard lifetime (70 years) drinking water ingestion rate of 2 liters per day containing 0.5 ng PCB per liter, a daily PCB exposure for a 70 kg adult of 0.014 ng/kg/day is estimated. Using the PCB cancer unit risk factor of 7.7 $(mg/kg/day)^{-1}$ (Exhibit A), an estimate of excess cancer from this exposure route of 1.0×10^{-7} is obtained, under worst case conditions. This cancer risk is well below the 10^{-6} to 10^{-5} range of risk generally considered acceptable by regulatory agencies.

38.4.2.2 Estimates of Direct Contact Exposures

As discussed in Section 2.7.1, a 1983 survey (Hite, 1984) of lake sediments showed PCB levels of less than 0.01 to 0.27 mg/kg in a region next to the Area 9 Landfill. Other surveys (Ruelle and Adams, 1984) detected 0.41 and 0.76 mg/kg in the same bay region. Sediments from mid-lake north of the landfill, were below 0.05 mg/kg in the latter survey. Elevated PCB levels, presenting the opportunity for ingestion exposures by benthic organisms, are considered in the ingestion exposure discussion below. It is doubtful that PCBs bound to sediment are subject to dermal absorption by aquatic organisms but they might contribute to localized areas of elevated surface water concentrations which may be available for absorption.

38.4.2.3 Estimates of Ingestion Exposures

Human

As part of the field investigation, specimens of fish including bass, carp, bullheads, and catfish were collected from Crab Orchard Lake. The edible tissues of these species were analyzed for PCBs, pesticides, cadmium, lead, and mercury. The results, presented in Table 38-2 and Figure 38-3, indicate that the ingestion exposure route is complete for humans and wildlife consuming fish taken from certain portions of the lake. PCB residues were elevated primarily in sampling location #2C, with lower levels detected in locations #1B and #3C.

Only two of the samples (out of 29 analyzed) contained residues in excess of the FDA action level of 2 mg/kg. Both of these samples

consisted of carp taken from location #2C, near a general area previously found to contain PCB residues in lake sediments. Carp are a bottom-feeding species which would come into direct contact with contaminated sediments, as would other bottom feeders such as catfish. With the exception of the fish sampled at the control location #10J at the Crab Orchard Lake dam, all fish sampled were from portions of the lake which are not readily accessible to recreational fishing, and from which movement into the remainder of the lake is constrained by two rather narrow passages. Since neither bass, carp, bullheads, nor catfish are particularly migratory, it is unlikely that fish in the eastern end of the lake make a significant contribution to populations in the remaining portion of the lake.

This ingestion exposure risk analysis involves a 'worst case' conservative estimate based on an individual who relies upon fish from Crab Orchard Lake for his or her total fish diet over a 5, 10 or 70 year period. The risk levels presented in Table 38-3 assume that PCB residues in fish remain constant over the specified period of time. Each of these assumptions may be improbable. For instance, it is more likely that an individual would obtain some portion of his fish diet from other sources, including ocean fish, fish from nearby lakes or rivers, or fish from commercial fisheries, which would diminish the risk estimates proportionately. In addition, PCB residues in fish from Crab Orchard Lake are likely to decline over time, because of decreasing levels available for uptake from bottom sediments due to the natural sedimentation process in the lake and remediation of on-land potential contaminant sources, and continuing replacement of aquatic populations.

It is apparent from examining the range of risks presented in Table 38-3 for ingestion of fish under various dietary scenarios that the most sensitive variable in the risk estimate is the total quantity of fish captured from Crab Orchard Lake and consumed over a lifetime which contains the level of PCB fillet residues currently observed. The upper bound cancer risk estimate of 2.0×10^{-3} represents the consumption of approximately 30 g of catfish (which are bottom feeders) from Crab Orchard Lake every day during a 70 year lifetime (sports fisherman scenarios, Table 38-3). This estimate assumes that 95 percent of the catfish are taken from the western area of Crab Orchard Lake, and 5 percent are taken from the east, since the fishable acres in the east end comprise about 3 percent of the total 7,000 fishable acres in Crab Orchard Lake. In addition, due to its average water depth of 2-3 ft., the east area is inaccessible to boat fishing and therefore fewer fish would be taken from the east side of the lake. More realistic consumption scenarios for persons who obtain their entire fish diet solely from Crab Orchard Lake (if any) may be those risk levels derived for periods of 5 and 10 years, with associated risks between 10^{-5} and 10^{-4} (one in one hundred thousand to one in ten thousand).

Cancer risks can also be presented in terms of the number of meals of fish from Crab Orchard Lake. For individuals consuming fish from the western portion of the lake, about 40 to 80 meals of mixed fish, assuming an average meal size of 0.5 to 0.25 lb and a catch similar to the creel census, would result in a lifetime excess cancer risk of about 10^{-5} (one in a hundred thousand). Therefore, if an individual wished to limit their lifetime excess cancer risk to one

TABLE 3B-3
CRAB ORCHARD LAKE

ESTIMATED RISK TO HUMANS
DUE TO CONSUMPTION OF FISH TISSUE

ASSUMPTIONS:
- 100 % of fish diet is captured at Crab Orchard Lake.
- Consumption of Crab Orchard fish continues over a 70-year lifetime, or during a 10-year or 5-year period.
- Undetected values are calculated as one half the analytical detection limit (0.2 mg/kg for RI data).
- Cancer unit risk factor of 7.7 $(mg/kg/day)^{-1}$ for Aroclor 1260 (°)
- East/West division denoted by Wolf Creek (200 fishable acres on eastern area, 7000 fishable acres total for lake.)

SCENARIO	AVERAGE PCB CONCENTRATION (mg/kg ww) (1)	(2)	<------ RISK LEVELS* ------> 70-year Lifetime	10-year Exposure	5-year Exposure
I. Average Fisherman (6.5 g fish per day or 10-20 meals per year)					
a) 95% West Bass/5% East Bass	0.34	0.20	2.5E-04	3.5E-05	1.8E-05
b) 100% West Catfish/Bullhead	0.32	0.08	2.3E-04	3.3E-05	1.6E-05
c) 95% West /5% East Catfish	0.61	0.51	4.3E-04	6.1E-05	3.0E-05
d) Creel Census (4)	0.30	--	1.7E-04	2.4E-05	1.2E-05
e) National Average (5,#) 1976-1979	0.29	--	2.1E-04	3.0E-05	1.5E-05
f) National Average (6,#) 1980-1981	0.18	--	1.3E-04	1.8E-05	9.2E-06
II. Sports Fisherman (30 g fish per day or 50-100 meals per year)					
a) 95% West Bass/5% East Bass	0.34	0.20	1.2E-03	1.6E-04	8.3E-05
b) 100% West Catfish/Bullhead	0.32	0.08	1.1E-03	1.5E-04	7.4E-05
c) 95% West /5% East Catfish	0.61	0.51	2.0E-03	2.8E-04	1.4E-04
d) Creel Census (4)	0.30	--	7.8E-04	1.1E-04	5.6E-05
e) National Average (5,#) 1976-1979	0.29	--	9.6E-04	1.4E-04	6.8E-05
f) National Average (6,#) 1980-1981	0.18	--	5.9E-04	8.5E-05	4.2E-05

NOTES & REFERENCES:

(1) Averages are calculated assuming fish without detected PCB residues contain such residues at one half the analytical detection limit.
(2) Averages are calculated assuming fish without detected PCB residues are free of such residues.
(3) Derived using a 1976 Creel Census survey and average concentrations in fish species detected in the RI and in monitoring studies conducted by the State of Illinois (see Section 2.7). Based on the Creel Census data, the relative catch per boat expedition at Crab Orchard Lake is comprised of roughly, 35% bass, 31% bluegill sunfish, 14% catfish, 12% crappie and 8% bullhead.
(4) ATSDR (November, 1987). Draft Toxicological Profile on PCBs.
(5) Schmidt, CJ et al. (1985). National Pesticide Monitoring Program. Arch. Environ. Contam. Toxicol.; 14:225-60.

(#) Fillet residues calculated as one third reported whole body residue.
(°) The potency factor of 7.7 $(mg/kg/day)^{-1}$ is based on studies using Aroclor 1260; only Aroclor 1254 residues were detected at Crab Orchard Lake. Available data neither demonstrate nor preclude the carcinogenicity of Aroclor 1254.
(+) Additive risks due to PCB/TCDF residues in fish might be obtained by adding 15 percent to risk level noted for PCBs.

chance in one hundred thousand, the total number of meals consumed in a 70-year lifetime from Crab Orchard Lake (western side excluding large channel cat) should be in the range of 40 to 80 meals. From a practical standpoint, four to eight meals annually over a ten year period would result in acceptable levels of risk similar to other dietary risks. For a person only occasionally visiting the lake, a higher rate of fish consumption would not yield unacceptable risks.

Stuart et al. (1984) presented the results of analyses of Crab Orchard Lake fish for polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). These were whole fish samples of largemouth bass, channel catfish, and carp captured from the eastern portion of the lake. The most highly toxic of this class of chemicals, 2,3,7,8-tetrachlorodibenzodioxin (TCDD), was not detected. One catfish contained unspecified penta- and hexachlorinated PCDD. Some of the fish analyzed contained 2,3,7,8-tetrachlorodibenzofuran (TCDF) at a mean level of 15 ppt, and two fish of the ten analyzed contained other TCDF isomers, with a group average of 2.5 ppt. Clark (1988) has indicated that in the environment, the 2,3,7,8-TCDD and TCDF compounds, the more toxic congeners of these classes, appear to preferentially bioaccumulate. This might in part explain why other TCDF congeners present in commercial PCBs were not detected in this fish survey.

Fink (1986) uses a conversion factor of 1/3 to estimate fillet concentrations of bioaccumulated organic contaminants to fillet concentrations. Therefore, the whole body TCDF data are converted to 5 ppt of 2,3,7,8-TCDF and 0.8 ppt of other TCDFs in fillets. Under the upper bound fish consumption scenario constructed for

PCBs, and assuming lifetime daily consumption of 30 grams catfish with no difference in PCDF concentration between the east and west portions of the lake, a daily adult (70 kg) intake of 2.1×10^{-9} mg/kg/day of the 2,3,7,8-TCDF and 0.36×10^{-9} mg/kg/day of other TCDFs is estimated.

In the absence of an extensive toxicological data base for PCDFs in general, U.S. EPA (1987, 1988) has adopted a weight of evidence approach to risk assessment of these compounds, termed the toxic equivalency factor approach, as an interim risk assessment procedure. In this approach, available test data and quantitative structure-toxicity relationships have been reviewed in order to rank the relative toxicity of the various PCDD and PCDF compounds and congeners relative to the best characterized and most toxic compound of these groups, 2,3,7,8-TCDD. Under this approach, the 2,3,7,8-TCDF congener is given a toxicity equivalence factor of 0.1 relative to 2,3,7,8-TCDD, and all other TCDFs are given a factor of 0.001. The cancer potency factor of 2,3,7,8-TCDD is currently given as 1.56×10^{5} $(mg/kg/day)^{-1}$. Therefore, the potency factors of 2,3,7,8-TCDF and other TCDFs are estimated as 1.56×10^{4} (mg/kg/day) and 1.56×10^{2} $(mg/kg/day)^{-1}$, respectively. The lifetime cancer risks from exposure to these compounds at the above estimates of exposure are, consequently, 3.3×10^{-3} for 2,3,7,8-TCDF and 5.6×10^{-7} for the other TCDFs. Since the risk estimate for PCB exposure at the same fish consumption rate is 2×10^{-3}, the TCDF exposure might increase overall risk to 2.3×10^{-3}, a factor of 15 percent. This incremental risk factor could be added to all risk

estimates presented in Table 38-3 to give an upper bound additive risk for PCB and TCDF exposure.

In view of the inherent uncertainty in quantitative carcinogenic risk assessment, the biological significance of this estimated incremental risk is unclear. In addition, some of the PCDD-like congeners present in aroclor mixtures often used in toxicological evaluations of PCBs may be responsible for some portion of the observed toxicity (Hileman, 1988; U.S. EPA, 1987), depending on the effect studied and the biochemical mechanism involved. If this applies to this risk assessment, the incremental risk from TCDF exposure via fish consumption may be somewhat overstated. However, in view of the possibility that certain PCDD and PCDF congeners may preferentially accumulate in sediments and in fish, including the more toxic 2,3,7,8-tetrachloro derivatives, environmental mixtures might be enriched in these components relative to commercial mixtures. Therefore, a conservative approach to this aspect of the risk assessment is desirable.

Short term consumption of catfish containing the average PCB levels (Scenario C. of Table 38-3) observed at Crab Orchard Lake does not result in intake levels which would be a cause for concern from non-carcinogenic PCB toxicity. Using a threshold exposure level for short-term non-carcinogenic effects of 1 mg/kg/day and a safety factor of 100, U.S. EPA (1986) established a short term acceptable intake level of 0.7 mg/day for a 70 kg adult for a ten day exposure period. At a fish ingestion rate of 250 g/day, assuming one meal of catfish daily during ten consecutive days, and a fishing pattern of 95 percent of catfish obtained from the west end and 5 percent of catfish

from the east end of Crab Orchard Lake, a daily PCB intake of 0.61 mg/kg x 0.25 kg/day or 0.15 mg/day is estimated. This intake is well below the acceptable level of 0.7 mg/day for short term exposures..

For longer term exposures, ATSDR (1987) and U.S. EPA ECAO-CIN-414 (1987) have identified a no observed adverse effect level (NOAEL) for Aroclor 1016 for developmental toxicity (reduced birth weight) of 0.01 mg/kg/day upon chronic oral exposure in monkeys. U.S. EPA ECAO-CIN-414 (1987) established a long-term health advisory Reference Dose for non-carcinogenic effects by applying a 100-fold uncertainty (safety) factor to this NOAEL, or 0.0001 mg/kg/day. At an ingestion rate of 30 g fish per day x 0.61 mg PCB/kg fish, a daily exposure rate of 0.00026 mg/kg/day for a 70 kg woman is estimated. This upper bound exposure estimate of 0.00026 mg/kg/day is higher than the Health Advisory intake level. However, this exposure level does not necessarily imply that developmental toxicity will be observed in humans exposed at this level because of the conservative nature of the assumptions and safety factors used. In fact, this exposure level is 50 percent lower than the lowest observed adverse effect level (0.0004 mg/kg/day LOAEL) seen in the monkey study with a 100-fold uncertainty factor applied. In addition, the rhesus monkeys studied were more sensitive to PCB effects than another highly sensitive species, such as the mink.

To place these estimates in perspective, and to illustrate the sensitivity of the risk analyses to the average concentrations detected in fish (as well as to the procedure for analyses of fish samples), the risk levels associated with consumption of freshwater fish available in

interstate commerce in the U.S. were reviewed. The U.S. Fish and Wildlife Service reported that whole body PCB residues in freshwater fish nationwide in 1980-81 averaged 0.53 ppm (Schmidt et al., 1985). As an approximation, the edible portions of fish tissue are often assumed to contain one third the lipophilic contaminant level of whole fish due to decreased fat content (Fink, 1986). Thus, under the same conservative dietary assumptions used in this assessment (30 g of fish consumed every day over a 70-year lifetime) for an avid freshwater sport fisherman, it is estimated that a fillet PCB level of 0.53/3 or 0.18 mg/kg - the average level throughout the country in 1980-1981 - has an associated risk of 5.8×10^{-4}.

Similarly, the estimated risk level associated with consumption of freshwater fish containing the national average concentrations in edible tissue reported for the period between 1976 and 1979 is 9.6×10^{-4}, assuming the same lifetime consumption rate of 30 g fish daily (see Table 38-3). This risk estimate is proportional to the higher national average PCB concentration reported in ATSDR (1987) for the period between 1976 and 1979. The downward trend in residue content would appear directly related to decreases in general environmental PCB residues due to restrictions on the manufacture and use of PCBs. Risks to the average consumer of fish would be substantially less, as saltwater fish, containing much lower PCB residues than freshwater fish, comprise a significant portion of fish consumed in the typical diet.

Background risk levels are largely a function of the analytical protocol (and analytical detection limit) employed in the analysis of fish samples. This dependence is illustrated, in part, by the

magnitude of the above estimated risks associated with consumption of commercially available fish containing the nationwide average PCB concentration reported in ATSDR (1987). To illustrate the direct dependence of the risk levels estimated for this site to the detection limit of 0.4 mg/kg employed in this analytical program (a common detection limit utilized nationwide in the analysis of PCBs in fish tissue), the risks associated with consumption of only fish flesh with results below the detection level were estimated. Assuming the undetected results are averaged as being equal to the detection limit of 0.4 mg/kg, and are consumed at a rate of 30 g daily during a 70 year lifetime, the risk is estimated to be 1.3×10^{-3}. Thus, in order to provide a less biased while still conservative estimate, scientists and regulatory personnel often utilize one half the analytical detection limit as a proxy for the undetected results in computing the average of a particular data set. Using this procedure to re-evaluate the risks associated with consumption of fish samples which do not contain detectable PCB levels results in a risk level of 6.6×10^{-4}. Use of the preliminary data reported in recent studies by SIU (Kohler, undated), which revealed an average PCB concentrations of 0.2 mg/kg in fish tissue from Crab Orchard Lake, would yield comparable risk values as using one half of the detection limit of 0.4 mg/kg.

There are several factors in addition to the total quantity of fish consumed, the estimated average level of contamination, and the analytical detection limit which may have a direct bearing upon the nature of this analysis. These factors should also be considered when evaluating and using the above risk estimates:

1. It is well recognized that certain methods of cleaning and cooking fish can remove significant amounts of contaminants found in fatty portions of edible fish tissue, including PCBs (Cordle et al., 1982). For instance, mean whole body PCB residues in lake trout from Lake Michigan in 1973-1974 of 18.9 to 22.9 mg/kg were decreased to 1.03 - 4.67 mg/kg after cleaning (i.e. removing head and internal organs) and cooking. Zabik et al. (1979) also demonstrated a significant reduction of PCB levels in lake trout fillets containing large amounts of fat by various cooking techniques including broiling, roasting, and microwaving. The degree of residue removal was dependent on the cooking method, with broiling and roasting more effective than microwave preparation.

Skea et al. (1979) showed a lower degree of removal using less fatty fish; brown trout and small mouth bass. Smoking removed 12 percent of Aroclor 1254 residues in the brown trout, whereas broiling increased the residue level, presumably by removing tissue water. In the smallmouth bass, baking increased the residue level while deep frying reduced the level by nearly 50 percent. Zabik et al. (1982) compared several preparation methods for effects on PCB residue level in cooked and raw carp fillets and reported either no effect or an increase in concentration. Thus, the degree of residue reduction of PCBs by cooking seems dependent on the species concerned, primarily due to variation in fat content, and cooking methods used, due to varying efficiencies for removal of fat and tissue water.

The presented assessment assumes that all PCBs in the fillet before cooking are carried over in the cooked meal. It is quite possible, however, that some portion of the residues would be lost during preparation of the food by certain methods, although no experimental data were located for the species assessed (largemouth bass and catfish). This lack of specific data and the variability observed in the literature precludes a quantitative estimate of possible risk reduction by cooking. As a result, further investigation on the effect of preparation of these species would be a useful aid for further refinement of this site assessment.

2. As a conservative precaution, carcinogenic risks were estimated as if the PCBs in question were of a potency equivalent to Aroclor 1260, whereas only Aroclor 1254 residues have been observed in Crab Orchard Lake sediments and fish. Under the conditions of a National Cancer Institute (NCI) chronic bioassay of two-year duration, Aroclor 1254 did not demonstrate convincing evidence of carcinogenicity in rats (as reviewed in Exhibit C, Harbison et al., 1987). Because the experimental design of the NCI study may have lessened the power to detect elevated tumor incidence, U.S. EPA has assessed the risks of all PCB aroclors including 1254 as if these were all of equivalent potency to Aroclor 1260. Two studies of greater than two-year duration in rats with Aroclor 1260 produced elevated rates of liver tumors in the late stages of the animals' lives. As reviewed in U.S. EPA ECAO-CIN-414 (1987) and ATSDR (1987), the

Aroclor 1260 studies have been selected by regulatory agencies as the basis for carcinogenic risk assessment of all PCBs.

Based upon what is known about the comparative biochemical toxicology of PCB congeners, however, Aroclor 1254 may be less potent as a carcinogen than Aroclor 1260. The more heavily chlorinated PCBs (penta- and hexachlor-) are believed to be most toxic to the liver, and these congeners predominate in Aroclor 1260 relative to 1254 (ATSDR, 1987). Thus, even if Aroclor 1254 were to show carcinogenic properties in rats if tested under the same conditions as Aroclor 1260, it is possible that the effect would be less, due to lesser presence of highly chlorinated organics.

This argument concerning the potentially lesser potency of Aroclor 1254 may not be applicable to environmental exposures in all cases. Because of their relatively low water solubility and greater persistence, environmental Aroclor 1254 residues in sediments and fish may be enriched in heavily chlorinated congeners relative to the commercial mixture as manufactured. Thus, the environmental mixture of Aroclor 1254 congeners might on occasion more closely resemble toxicologically a commercial Aroclor 1260 mixture compared to a commercial 1254 mixture. This does not, however, appear to be the case for the fish residue data generated in the RI. Examination of the gas chromatograph scans by a PCB analytical specialist indicates that the RI Aroclor 1254 patterns were representative of standard Aroclor 1254, with little of no enrichment of the more heavily chlorinated congeners (Hill, 1988).

The study by Norback and Weltman (1985) which forms the basis for the quantitative potency estimate for Arocior 1260 and all other PCBs, is somewhat flawed and its relevance to human risk assessment has been questioned (Exhibit C, Harbison et al., 1987). The dose administered appears to have been hepatotoxic, excessive control mortality was observed, and a number of the animals which were scored as tumor-bearing had received a partial hepatectomy (a possible tumor promoting influence) earlier in the study. The liver tumors induced in treated animals appeared very late in the animals' lives, and did not appear to be life-threatening or metastatic. These observations inject additional uncertainty into the quantitative assessment of the health risks from ingestion of fish containing PCBs. Use of these data helps ensure, however, that the assessment of risks associated with PCB exposure errs on the side of overstating rather than understating the risks.

The factors discussed above, as well as those presented elsewhere in this report (Section 6.5 and Section 38.4.3, Analysis of Uncertainties), illustrate the necessity to consider the underlying uncertainty as well as the overall weight of evidence when applying this approach to a case-specific assessment.

Humans might also receive exposures to PCBs from consumption of certain duck and geese species which may have been exposed to residues from bottom sediments in the eastern area of Crab Orchard Lake. According to the Refuge Manager, the 43,000-acre Refuge supports a peak population of approximately 12,000 ducks and 100,000 geese during the fall and winter seasons. The duck population

consists of about 10,000 puddle ducks (surface feeders) and 2,000 diving duck species which spend an average of two months/year at the Refuge. Both geese and ducks utilize the eastern end of the lake for resting and loafing purposes. These species generally feed very little in the lake because farm crops are abundant throughout the Refuge. Diving ducks, although seen occasionally resting in the east end, generally prefer deeper water areas in the western portion of the lake. Although the number of waterfowl on the eastern end mudflats or shallows may reach 2-4,000 birds for several days, individual birds and subflocks are constantly changing with other groups elsewhere on the Refuge on a daily or hourly basis. Thus, an individual bird is estimated to spend no more than 10 days maximum per season in the eastern portion of Crab Orchard Lake. Based on this limited exposure to sediments in the east area of the lake, and the low residue levels present, ducks would not be expected to accumulate significant PCB residues. The ingestion of duck or geese from Crab Orchard Lake is not considered to represent a risk to humans or wildlife at the Refuge.

In summary, the estimated risks to humans associated with consumption of fish are within a range of 10^{-5} excess lifetime cancer risk for rates of fish consumption between 40 and 80 meals occurring over a few years or over a lifetime of 70 years. At higher rates of consumption (greater than a meal per month or 12 meals/year), for more than three consecutive years or a lifetime, excess cancer risks would exceed 10^{-5} for most consumption scenarios.

Wildlife

The presence of PCB residues in fish in Crab Orchard Lake creates a complete exposure pathway for piscivorous birds and mammals in this region. Mink, otters, and other fish-consuming mammals living near the eastern portion of the lake may be exposed to the levels of PCB residues which have been detected in sediments and bottom-feeding fish. In particular, mink are highly susceptible to the acute and subchronic effects of PCBs (Newell et al. 1987), and thus will be a focus of the wildlife risk assessment. Piscivorous and/or aquatic birds may also be exposed to PCB residues since such birds would be expected to obtain a portion of their total diet from captured fish or fish carrion taken from Crab Orchard Lake. As discussed below, the Refuge is an active nesting area for 2-3 pairs of bald eagles, an endangered species with documented sensitivity to chlorinated hydrocarbon pesticides and perhaps PCBs. Herons and osprey may also be exposed via ingestion of fish obtained from some portions of Crab Orchard Lake. Certain species of duck might obtain part of their diet from Refuge fish; however, according to the Refuge Manager, such species spend approximately two months per year at the Refuge as a temporary habitat during their excursion north, and, during this period, spend most of their time feeding or loafing in the deeper areas of the lake, on the west end of the Refuge. Less migratory species of duck and/or geese which might remain at the Refuge long enough to receive limited exposure to contaminants are predominantly vegetarian, and tend to feed on various agricultural crops surrounding Crab Orchard Lake. For the above reasons, duck and geese will not be considered in the quantitative assessment.

In order to assess risks to piscivorous mammals, it will be assumed that a population of mink exists on the eastern portion of Crab Orchard Lake. As reviewed in Newell et al. (1987), a typical mink diet consists of about 50 percent fish, and a 1 kg adult mink consumes approximately 150 g of food per day, or 75 g of fish. Fish in the eastern portion of the lake (17 samples, all species) contained an average of 1.2 mg/kg in the muscle fillet ("edible tissue"). Whole body residues may be reasonably assumed to be three times as high (Fink, 1986), or 3.6 mg/kg. Thus, a mink consuming 75 g per day of fish containing 3.6 mg/kg of PCBs would receive a daily exposure of 0.27 mg/kg body weight/day. Newell et al. (1987) estimated that a daily intake of 0.13 mg/day would provide a margin of safety of 5 fold against the lowest level of dietary PCB exposure associated with reproductive impairment in laboratory tests with mink. Thus, the worst case exposure scenario provides a safety factor of more than 3 fold over the lowest observed effect level in mink. In view of the observation that mink tend to feed predominantly on young fish which have not had the opportunity to acquire elevated contaminant levels (Ruelle, 1987), this safety factor is concluded to be adequate. Piscivorous mammals less sensitive than mink would be expected to be further below the effect threshold, because mink are the species with the greatest documented sensitivity to PCBs.

In contrast to a resident mink population, piscivorous birds on the Refuge can obtain food from a number of sources, including all of Crab Orchard Lake and other aquatic systems. In developing criteria for contaminants in fish flesh in regard to piscivorous wildlife, Newell et al. (1987) utilized a food consumption rate of 20 percent of body

weight per day for a number of piscivorous birds, including bald eagle, great blue heron, and osprey. An adult bald eagle body weight of 4.5 kg is used in the assessment below. According to the Refuge Manager, the peak winter population of eagles on the Refuge is 30 birds. If these winter eagles restricted their food source to the vicinity of Crab Orchard Lake, their diet would consist almost entirely of wounded waterfowl and miscellaneous road kills such as snakes, squirrels, and rabbits. During the spring and summer (May through August) only 2-3 nesting pairs of bald eagles utilize Crab Orchard Lake; the diet of these birds during these four months is made up of approximately 80 percent fish species and 20 per cent carrion. It is reasonable to assume that the total fish diet is taken randomly from all areas of the lake, given the large foraging range of this species. If sampling locations #1B, #2C, and #3G in Figure 38-3 are considered to be representative of the eastern area of the lake (east of Wolf Creek), and sampling locations #4H and #10J are considered representative of the species on the west side of the lake, the average fillet tissue concentrations of PCBs for all fish species from the west and east portions of the lake are 0.37 and 1.2 mg/kg respectively. (Similar to the assumptions for the human health risk assessment, samples with results below the detection level are considered as one half the analytical detection limit or 0.2 mg/kg). The relative volume of water in the west portion of the lake is greater, since the average depth of the western area is 8-9 ft. versus 2-3 ft. on the eastern side. In addition, the west area represents approximately 97 percent of the total fishable acres in Crab Orchard Lake. If it is assumed that the fish population is

roughly proportional to the area of the lake (3 percent east, 97 percent west), then the weighted average PCB fillet tissue concentration would be 0.39 mg/kg for the overall fish population in Crab Orchard Lake. The whole body concentrations consumed by eagles would be estimated to be three times this concentration (Fink, 1986) or 1.2 mg/kg.

Considering the above assumptions, a 4.5 kg adult eagle is estimated to consume 0.19 mg PCBs/kg/day during the four summer months when its intake is comprised of 80 percent whole fish and it maintains a consumption rate of 20 percent its body weight. Normalizing this intake over a one year period to be representative of both winter and summer dietary intakes, an exposure level of 0.057 mg/kg/day is estimated for eagles. Applying a 10-fold margin of safety to a no observed effect level (NOEL) for reproductive effects in PCB- exposed domestic hens (data on target species being unavailable), Newell et al. (1987) concluded that a daily intake of 0.11 mg/kg/day was an acceptable intake level for birds. The estimated daily intake (0.057 mg/kg/day) for eagles or other similar piscivorous birds at Crab Orchard Lake is roughly one half this value, and is well below the NOEL of 1.1 mg/kg/day for reproductive toxicity in domestic hens. Thus, for effects to be seen in eagles at this level, they would have to be significantly more sensitive to PCBs than domestic hens and the birds would have to obtain at least 80% of their fish diet from Crab Orchard Lake during both the summer and winter months.

Benthic and bottom feeding organisms may be at risk from ingestion of bottom sediments bearing PCBs, and possibly from

absorption of localized residues of these compounds in water near and beneath the sediment surface. However, these risks cannot be quantified due to lack of data on ingestion rates and bioavailability of these residues, and uncertainty regarding the ability of these compounds to produce acute and chronic effects at low levels. The risk levels evaluated above for piscivorous mammals and birds suggest that wildlife are not subject to unacceptable exposure levels due to contaminants in Crab Orchard Lake fish.

38.4.3 Sedimentation Analysis

As a followup to the Phase II investigation, split-spoon sediment core samples were collected from five locations (32-98, 103, 104, 105, 107, see Figure 36-3) within the embayment adjacent to the Area 9 Landfill. One core was also collected at a location (7E, 34-22) toward the center of the lake, which is further removed from surface discharges and is close to a narrow opening between the west and east lake portions where the water velocity would be greater. The depth to naturally occurring sublake soils was determined for each sediment core by noting the appearance, soil type, and moisture content at 6 in. to 1 ft. intervals for the continuous split-spoon samples. The depths of sediments were measured as 66, 51, 54, 33, and 54 inches respectively, or a mean of 51.6 inches for the Area 9 embayment. The depth for the sediment core from Location 34-22 at the center of the lake was 18 inches. Since the lake has existed for approximately 45 years, an average sediment deposition rate of 1.2 inch per year or 2.9 cm per year is estimated for the embayment, and 0.4 inch or 1.0 cm per year for the mid-lake areas. According to Fink (1987), these are deposition rates characteristic of productive reservoir systems.

Fisher et al. (1983) studied the release rate to overlying water of four polychlorinated biphenyls (tri- and tetrachloro-) contained in contaminated bottom sediments collected from Waukegan Harbor in Illinois. Having determined the rate of flux and the rate of migration through interstitial water in the sediments by molecular diffusion, these authors concluded that even the most mobile of the PCBs studied had an extremely low diffusivity through the sediments. Using these values for upward migration, it was determined that sediment deposition would overtake diffusion as the determining element for PCB release if other disturbing influences such as wave action, and biological disturbances are ignored. For instance, it was concluded that a sediment deposit rate of 0.004 in. or 0.01 cm per year would remove PCBs in a surface sediment layer from communication with overlying water in approximately three years.

In the absence of other factors, therefore, the estimated sediment deposition rate for the bay would remove sediment containing PCB residues from communication with the free water column within less than a year once remediation of on land contaminants has been completed. As noted by Fisher et al. (1983), sediment disturbances such as caused by wind or current, or by fish and benthic organisms could retard this rate of encapsulation of contaminants. Since this region of the lake is frequented by carp and other bottom-feeding fish, it may be more conservative to assume that the upper 5 cm or so of the sediments, constituting the active sediment zone, is being continuously mixed by the action of fish, waves, and benthic organisms. Thus an influx of 2.9 cm of non-contaminated sediment per year into the bay would reduce existing PCB concentrations by about 60 percent annually, by simple dilution, exclusive of biodegradation or other dissipative mechanisms. Thus, three years after

the transport of PCBs to the bay ceased, residues would be reduced to about 6 percent of previous values, and to 1 percent after 5 years.

38.4.4 Analysis of Uncertainties

As has been discussed elsewhere in this report (see above and Section 6.5), several key areas of uncertainty are contained in this risk assessment. To ensure that the assessment does not understate the magnitude of the upper bound of risk, multiple ranges of assumptions (i.e. dietary habits, residue levels, duration of exposure) spanning a possible least case scenario to a possible worst case scenario were employed. The approach used to derive the worst case risk estimates is not intended to estimate the actual risks which may be associated with this site, which are most likely defined by a probability distribution rather than a discrete value. The probability that even a small subset of the human and wildlife populations might begin to meet the set of worst case assumptions used is small. It is unlikely that the level of risks calculated could be detected above background incidence of disease. The upper bound risk estimates are of importance in this case because of the severity of the potential effects, not because they are likely to occur. In addition, the quantitative assessment model assumes that all PCB aroclors are as carcinogenic as the most toxic Aroclor 1260, even though there is reason to conclude that Aroclor 1254, which was the only Aroclor detected in the residues at this site, may be less potent.

Additional uncertainty is introduced due to the limited size of the analytical sampling program for this RI; this degree of uncertainty must also be considered in the interpretation and conclusions derived from the evaluation of this site. Although limited, the RI analytical data are

generally supported by the more extensive database developed as part of ongoing monitoring programs conducted by the Illinois Department of Public Health (IDPH, 1976-1987) and SIU (Kohler, undated). The results from these monitoring data are presented in the Review of Previous Investigations in Section 2.7.

38.5 Preliminary Remedial Alternatives

Three samples of lake waters remote from the Refuge waterworks intake contained part per trillion levels of PCBs; these results were above the Ambient Water Quality Criteria for human health, but only one sample exceeded the concentration standard for protection of aquatic life. Lake waters near the waterworks intake and after treatment did not contain detectable levels of PCBs (detection limit of 0.005 ug/L). The treated water supplies exceeded the criteria for some trihalomethane compounds, likely due to the drinking water requirements for residual chlorine after treatment. Corrective measures have been instituted in the treatment of these waters to reduce trihalomethane level below the standards. In addition, the water samples from some areas of Crab Orchard Lake contained low levels of arsenic and mercury above the ambient water criteria for human health. However, all detected constituents were below the Ambient Water Criteria for protection of aquatic life with the exception of PCBs in one sample from the embayment near Area 9. Sediment samples from the lake contained phthalates and trace semi-volatile compounds; however, the phthalates were also detected in the laboratory blanks. The sediment samples from the Area 9 embayment also contained microgram (ppb) levels of PCBs.

As evaluated in the previous section, the effects of contaminants in lake sediments appear to have resulted in PCB concentrations in some fish from Crab Orchard Lake above the FDA tolerance level for this regulated substance. Two

carp composites (6.4 and 3 mg/kg PCBs; 3.9 and 4.4 mg/kg when re-analyzed) and one bass composite (1.05 mg/kg mercury) collected from the eastern area of the lake exceeded the FDA criteria of 2.0 mg/kg for PCBs and 1.0 mg/kg for mercury.

These data were evaluated in a detailed risk assessment (Section 38.4), however, with the conclusion that the risks to local fishermen and to wildlife do not warrant direct remediation of lake sediments. Based on the review of previous information and the evaluation of current site conditions as part of this investigation, no remedial measures are recommended for Crab Orchard Lake, except for periodic monitoring of lake waters, as detailed in Attachment 1. The attached Monitoring Plan proposes continuing monitoring of waters at this site and other tributaries and drainage routes to Crab Orchard Lake.

Periodic monitoring of lake fish and sediments is also recommended, whether as part of a continuing program conducted by the State of Illinois, or in conjunction with the proposed plan in Attachment 1. Further monitoring of fish residues with a method permitting a lower analytical detection limit would lead to a more accurate quantitative determination of risks associated with consumption of fish containing fillet PCB residues at the lower end of the range observed in the various investigations, in terms of both species analyzed and area of the lake sampled. In addition, any such monitoring should include analyses for PCDD and PCDF congeners in fillet tissue to more accurately determine overall risk from fish consumption.

38.6 Conclusions and Recommendations

It can be concluded that the waters of Crab Orchard Lake generally meet all regulatory standards and criteria for human health and aquatic life with the exception of low concentrations of PCBs, arsenic and mercury in isolated

locations. Some sediments contained low levels of PCBs, phthalates, and other organics.

Given the circumstances of Crab Orchard Lake, including observed fishing patterns, the remedial measures contemplated for various potential PCB sources around the lake, the low levels and limited areas of PCB contamination in lake sediments, the apparent health of the existing aquatic ecosystem, and the natural self-sealing mechanism provided by the lake sedimentation, no remedial measures are recommended for the lake. For these reasons, the lake will not be carried forward into the Feasibility Study.

As with several other sites not being evaluated in the FS, a proposed followup monitoring program for lake waters has been included as Attachment 1 of this report. It is further recommended that lake fish and sediments continue to be monitored on a periodic (e.g. annual) basis following implementation of source control measures.

SECTION 39 - SITE 35, AREA 9 WATERWAY

39.1 Site Description

Site 35 is a low lying spot in an agricultural field to the east of Area 9. (See Figure 39-1). Vegetation does not grow in the area of depression, indicating the potential presence of contaminants.

39.2 Site Investigations

39.2.1 Phase I Site Investigations:

One composite soil sample was collected during Phase I.

39.2.2 Phase II Site Investigations:

No samples were collected during Phase II.

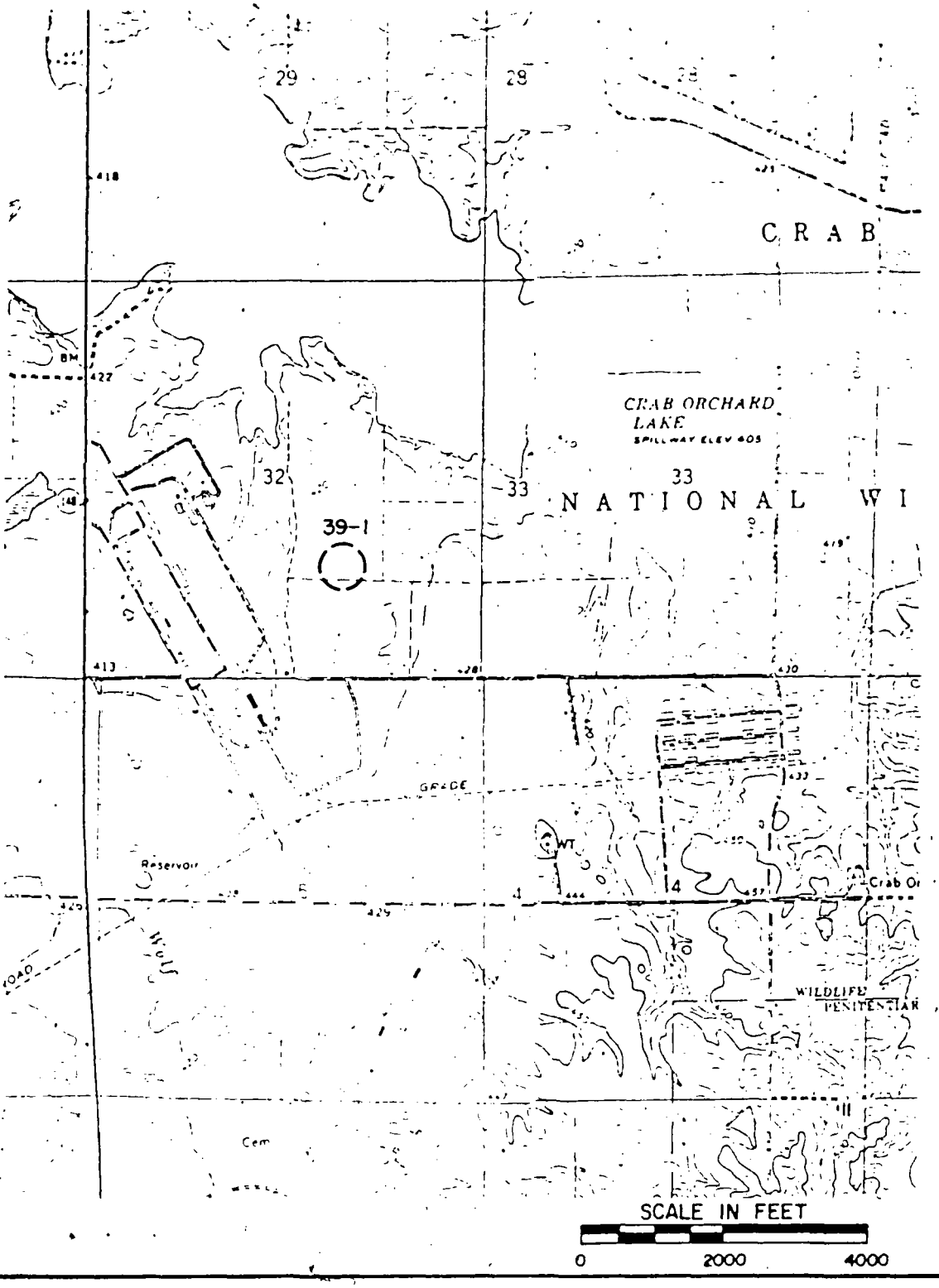
39.3 Analytical Results (See Appendix I, Page 31)

Unusually high specific conductance, 11,650 umhos/cm, was measured in the soil sample, possibly due to the accumulation of salt. A trace (0.016 mg/kg) of PCBs was detected. The concentrations of all other parameters were below those detected at the Refuge control sites, although the volatile and semi-volatile organics data for Phase I are questionable due to QA/QC deficiencies or unreliable support data (see Exhibit B). Some compounds reported as not detected may in fact be present.

FIGURE 39-1

SITE 35
AREA 9 WATERWAY

PHASE I



O'BRIEN & GERE ENGINEERS INC

F 1 1

39.4 Environmental Effects

39.4.1 Qualitative Assessment

This site was chosen for investigation based on an inspection of the site by the Refuge Manager. The lack of vegetation in the depression of this site led to speculation that this area might be contaminated.

Other than an unusually high specific conductance, which was attributed to the salt content of the soil, all other parameters were within the range of their respective Refuge background levels. Aerial photos show this location to be a wet spot within a low-lying area of an agricultural field. It is speculated that agricultural runoff containing fertilizers, etc. accumulates at this location and then evaporates, leaving behind an accumulation of salts.

Because there is no established waste source at this location, it is not possible to have a "complete" exposure scenario. Therefore, on the basis of the information generated, it can be concluded that the site does not represent a risk of chemical exposure to potential human or wildlife receptors.

39.4.2 Quantitative Assessment

Because a complete exposure scenario could not be identified in the qualitative assessment, there is no basis for preparing a quantitative risk evaluation.

39.4.3 Analysis of Uncertainties

The major information relied upon for evaluating this location was a site inspection and a sample analysis. An inspection of the site revealed a depression in the field that lacked vegetation. It was speculated that contaminants may have caused the lack of vegetation.

Chemical residue information consisted of analytical results for one surface soil sample. This information was obtained only for the top one foot of soil; deeper soil borings were not conducted. Since only a trace of PCBs were detected and all other parameters were below detection limits, there is no evidence to suggest that this site is contaminated.

It can be concluded that the data generated are adequate for evaluation of this site. The sampling analyses indicate that the site does not contain contaminants at levels that would be detrimental to human health or to the environment.

39.5 Preliminary Remedial Alternatives

Preliminary Phase I screening results discussed in the previous sections indicated that this site does not contain contaminant levels that would result in a negative environmental impact. Therefore this site was not included in the Phase II investigations. There will be no further evaluation of remedial alternatives, and this site will not be included in the FS.

39.6 Conclusions and Recommendations

It can be concluded that the East Waterway site does not represent a chemical exposure risk to human or wildlife receptors at the Refuge or at other locations. No further evaluation is recommended for this site.

Respectfully Submitted,

O'BRIEN & GERE ENGINEERS, INC.

C.B. Murphy Jr.

Cornelius B. Murphy, Jr., Ph.D.
Senior Vice President
O'Brien & Gere Project Officer

Prepared by:

Henry T. Appleton, Ph.D., Senior Toxicologist
Donald T. Bussey, Hydrogeologist
Steven R. Garver, P.E., Vice President
David R. Hill, Manager, OBG Laboratories
Swiatoslav W. Kaczmar, Ph.D., Managing Toxicologist
Cynthia M. Klevens, Project Engineer
Mark LaRue, Researcher
Richard G. Stromberg, Sr. Project Hydrogeologist
Matthew Traister, Research Engineer

REFERENCES

Adams, N.R. (FWS). Memorandum to Project Manager, Crab Orchard NWR. Chemical Contamination on Crab Orchard Refuge. (Oct. 13, 1982). 41

Adams, W. Memorandum to Dr. James Edler. Hazardous Waste Sites on Service Lands. (Oct. 17, 1983). 29

Adams, Wayne D. (U.S. DOI). Memorandum to John Ellis, Twin Cities. PCB analyses - Fish Samples in Crab Orchard Lake. (May 30, 1984). 21

Adams, Wayne D. (U.S. DOI). Memorandum to James Gritman, U.S. EPA. Sangamo Electric - Crab Orchard NWR. (May 24, 1984). 23

Adams, Wayne D. (U.S. DOI). Memorandum to John Ellis. Olin's Testing for PCBs in the Sangamo Plant Area. (Jul. 17, 1984). 12

Adams, Wayne D. (U.S. DOI). Memorandum to D. Stalling, CNFRL. Soil and Water Samples - Sangamo - Marion. (Aug. 21, 1984). 8

Agency of Toxic Substances and Disease Registry (ATSDR), Acting Director, Office of Health Assessment, Memorandum to Ms. L. Fabinski, Public Health Advisor, EPA Region V, June 12, 1986. Review of Raw and Finished Water Data, Crab Orchard National Wildlife Refuge (SI-86-097).

American Association of Petroleum Geologists. Geological Map. 1965.

Arnett, C. Ray (U.S. FWS). Letter to Senator Alan Dixon. Presence of PCBs in Crab Orchard Lake Ecosystem. (May 22, 1984). 50

ATSDR (1987). Draft Toxicological Profile for N-nitrosodiphenylamine. October 1987.

ATSDR (1987). Toxicological Profile for Selected PCBs. Agency for Toxic Substances and Disease Registry. Draft. November 1987.

Bacci, E. and Gaggi, C. (1985). Polychlorinated Biphenyls in Plant Foliage: Translocation or Volatilization from Contaminated Soils? Bull. Environ. Contam. Toxico. 35:673-681.

Bandiera, S., Sawyer, T., Romkes, M., Zmudzka, B., Safe, L., Mason, G., Keys, B., and Safe, S. Polychlorinated dibenzofurans (PCDFs): effects of structure on binding to the 2,3,7,8-TCDD cytosolic receptor protein, AHH induction and toxicity. Toxicol. (1984).32:131-144

Bell, Charles R. (Illinois EPA). Memorandum to Ira Markwood. Crab Orchard Lake Sampling. (May 25, 1984). 22

Bell, Charles R. (USEPA, FOS-DPWS, Springfield). Memorandum to Joseph E. Stuart, U.S. EPA, FOS-DPWS, Marion. Williamson County - Crab Orchard Refuge PWS; Facility No. 1997037. (Aug. 8, 1984).

Beyer, W.N., Chaney, R.L., and B.N. Mulhern. 1982. Heavy Metal Concentrations in Earthworms from Soil Amended with Sewage Sludge. J. Environ. Quality. 11:381-385.

Boice, Richard (USEPA Region V). Letter to R. Ruelle, U.S. FWS. Phase I Data QA/QC comments, Crab Orchard RI. (Feb. 18, 1987).

Bradlaw, J.A., Garthoff, L.H., and Hurley, N.E. Comparative induction of aryl hydrocarbon hydroxylase activity in vitro by analogues of dibenzo-p-dioxin. Fd. Cosmet. Toxicol. (1980). 18:627-635.

Bruckner, J.V., Khanna, K.L and Cornish, H.H (1974). Effect of Prolonged Ingestion of Polychlorinated Biphenyls on the Rat. Fd. Cosmet. Toxicol. V12:323-330.

Bush, B., Shane, L.A., Wilson, L.R. Barnard, E.L. and Barnes, D. (1986). Uptake of Polychlorinated Biphenyl Congeners by Purple Loosestrife (Lythrum salicaria) on the Banks of the Hudson River. Arch. Environ. Contam. Toxicol. 15:285-290.

Byram, Scott. Memo to File. Crab Orchard National Wildlife Refuge - Sangamo Dump. TDD No. R5-8308-6. (Sep. 21, 1983). 31

Byram, S. Ecology & Environmental. Potential Hazardous Waste Site - Site Inspection Report. (Oct. 3, 1983). 48

Callahan, M.A. et al. Water Related Environmental Fate of 129 Priority Pollutants, VI. EPA-440/4-79-0292. (Dec., 1979).

Carlson, Richard J. (IEPA). Letter to Wayne Adams, FWS. Notice of Sealing of Site. (May 16, 1984). 24

CDC/ATSDR (1987) Toxicological Profile for Selected PCBs. Agency for Toxic Substances and Disease Registry. Draft. November 1987.

Chew, R.M. (1965). Water Metabolism of Mammals. In Mayer, W.V. and R.G. VanGelder. Physiological Mammalogy, V.2. pp. 43-178. Academic Press. New York.

Chu, C.K., Stella V.J. Bruckner, J.V. and Jiang, W.D. (1977). Effects of Long-Term Exposure to Environmental Levels of Polychlorinated Biphenyls on Pharmacokinetics of Pentobarbital in Rats. V.66, N.2:238-241 (February, 1977).

Clement Associates, Inc. (1985). Chemical, Physical, and Biological Properties of Compounds present at Hazardous Waste Sites, Final Report to U.S. EPA, Arlington, Va. (Sep. 27, 1985).

Contract Laboratory Program for Organic Analysis - Multimedia, Multicomponent, see USEPA citations.

Cordle, F., R, Locke, and J. Springer. 1982. Risk Assessment in a Federal Regulatory Agency: An Assessment of Risk Associated with the Human Consumption of some Species of Fish Contaminated with Polychlorinated Biphenyls (PCBs). Environ. Health Perspec. 4:171-182.

Daniels, W. and Kramkowski, R. (1986). Health Hazard Evaluation Report, National Institute for Occupational Safety and Health. HETA 85-334-1676, Olin Corp., Marion, Illinois. (Mar., 1986).

Davis, S.N. and Dewiest, R.J. (1966). Hydrogeology. John Wiley & Sons.

Dawson, J.P. (Olin Corporation). Memorandum to L. A. Krause. Chlorinated Biphenyls - Marion, Wipe Samples. (Aug. 7, 1984).

Deifenbach, Russ (Dept. of Health and Human Services). Memorandum to Louise A. Fabinski, EPA Region V. Occupational Health Hazard - Olin Corporation, Crab Orchard NWR, Marion, Illinois. (Apr. 14, 1985).

Envirodyne Engineers, Inc. PCB Profile of "I" Area. Prepared for Olin Corporation. (Aug. 1984).

Envirodyne Engineers, Inc. PCB Profile of Three Locations in Area "I". Prepared for Olin Corporation. (Aug. 1984).

Fink L. (1987) Telephone conversation between Larry Fink, Chemist, USEPA Region V, Grosse Isle, MI, and Henry Appleton, Senior Scientist, O'Brien & Gere, Syracuse NY. Nov. 3, 1987.

Fink L. E. (1986). Risk Assessment Calculations for PCBs in Crow Island Dredged Material. Great Lakes National Program Office, Grosse Isle MI. Oct. 1986.

Fisher J.B., R.L. Petty and W. Lick. (1983). Release of Polychlorinated Biphenyls from Contaminated Lake Sediments: Flux and Apparent Diffusivities of Four Individual PCBs. Environ. Pollution (series B) 5:121-132.

Forbis, Leanne (Analytical Bio-Chemistry Laboratories, Inc.). Letter to Glen Clarida, U.S. EPA-DLPC, Marion, Illinois. PCB Analysis in Catfish. (Jun. 15, 1984). 16

Frankland, Les. Memorandum to Bob Cole. Transmittal of Results from 1981 Deer Sampled at Crab Orchard Lake Wildlife Refuge. 51 samples. (Jul. 9, 1984). 54

Frankland, Les (Illinois DOC). Memorandum to Matt Rice, FWS. Crab Orchard Lake Synopsis. (Aug. 21, 1984). 46

Frye, J.C. Outline of Glacial Geology of Illinois and Wisconsin, Quaternary of the United States, Princeton, NJ. (1965)

Gifford, Michael A. (Ecology & Environmental, Inc.). Memorandum to File. Illinois/TDD R5-8308-06A; Crab Orchard National Wildlife Refuge/Sangamo Dump. (Apr. 5, 1984). 27

Gritman, James C. Letter to Dr. David Kenney. Results of Contaminant Analyses on Soil and Fish Samples from Crab Orchard National Wildlife Refuge, sampled Sept. 1982 (No date). 56

Gritman, James C. (U.S. FWS, Twin Cities, Mn). Letter to Valdas V. Adamkus, USEPA, Chicago, IL Hazardous Waste Site at Crab Orchard NWR. (Jun. 1, 1984). 18

Gross, M.L. (Univ. of Nebraska). Letter to R. Ruelle, FWS. PCDD and PCDF Data. (Aug. 26, 1983). 33

Gross, M. (Univ. of Nebraska). Analysis of Polychlorodibenzo-p-dioxin (PCDD) and Polychlorodibenzofuran (PCDF) in Soil Samples. (Oct. 7, 1983). 30

Guyton, A.C. (1947). Measurement of the Respiratory Volumes of Laboratory Animals. American Journal of Physiology 150:70-77.

Harbison, R.D., R.C. James, and S.M. Roberts. (1987). Biological Data Relevant to the Evaluation of Carcinogenic Risk to Humans. Prepared for the Scientific Advisory Panel, Safe Drinking Water Enforcement Act, State of California. (August, 1987). University of Arkansas School of Medicine, Little Rock, Arkansas. See Exhibit C.

Hawley, J.K. (1985). Assessment of Health Risk from Exposure to Contaminated Soil. Risk Analysis V.5:289-302.

Hileman, B. (1988). The Great Lakes Cleanup Effort. Chemical & Engineering News. Washington D.C. (February 1988).

Hill, DR. (1988). Conversation with H.T. Appleton, O'Brien & Gere, Syracuse, NY, May 31, 1988.

Hite, R. L. and King, M. Biological Investigation of the Crab Orchard Creek Basin, Summer 1975. Illinois Environmental Protection Agency. (Apr. 1977). 44

Hite, Robert L. (EPA-DWPC). Letter to Richard Ruelle, FWS. PCB Monitoring in Crab Orchard Lake. (May 8, 1984). 25

Hite, Robert L. and Martin H. Kelly. Staff Report. Polychlorinated Biphenyl Monitoring, Crab Orchard Lake, 1983. (May 1984). 5

HSDB (1987) Hazardous Substances Data Bank. N-nitrosodimethylamine. National Library of Medicine. Bethesda, MD.

Huckins, Jim. (U.S. DOI-FWS). Letter to Wayne Adams, CONWR. Results of Crab Orchard Sample Analyses from Area 11. (Sep. 18, 1984). 45

Hurley, J. (IEPA, DPWS). Report to J. B. Tolson (U.S. Fed. Penitentiary - Marion). Trihalomethane Analysis Report. (May 8, 1984). 51

Hwang S.T., Falco J.W. and Nauman C.H. (1986). Development of Advisory Levels for Polychlorinated Biphenyls (PCBs) Cleanup.

ICF Incorporated. (1985). Superfund Health Assessment Manual, Draft, Washington, D.C., EPA Contract No. 68-01-6872. (May 22, 1985).

Illinois DOA. Memorandum to Les Frankland, IL DOC. Toxicology Report on Fish. (Apr. 4, 1983). 36

Illinois Department of Public Health Monitoring Data for Crab Orchard Lake (1976-1987). *Transmittal from R. Boice to O'Brien & Gere*, November 2, 1987.

Illinois Environmental Protection Agency. Staff Report. Biological and Water Quality Survey of Crab Orchard Creek in Vicinity of Marion Wastewater Treatment Plant, Marion, Illinois, 1979 and 1980. (Jul. 1981). 42

Kelly, M.H. and Hite, R.L. Chemical Analysis of Surficial Sediments from 63 Illinois Lakes, Summer 1979. Illinois Environmental Protection Agency. (1981). 43

Kelly, M.H. and Hite, R.L. Evaluation of Illinois Stream Sediment Data: 1974-1980. Illinois Environmental Protection Agency. (Jan. 1984).

Kenney, David (IL DOC). Letter to J. Gritman, FWS. Fish Sample Analysis Report for Crab Orchard Lake. (Apr. 18, 1983). 53

Kimbrough, R.D., Linder, R.E. Gains, T.B 91972). Morphological Changes in Livers of Rats Fed Polychlorinated Biphenyls. Arch. Environ. Health V.25:354-364:

Krause, L.A. (Olin Corporation). Memorandum to A. Heinz. Polychlorinated Biphenyl Study - Marion, Preliminary Report. (Jul. 20, 1984).

Linder, R.E. Gains, T.B., Kimbrough, R.D. (1972). The Effect of Polychlorinated Biphenyls on Rat Reproduction. Fd. Cosmet. Toxicol. V.12:63-77.

Lindsay, W.L. Chemical Equilibrium in Soils. John Wiley & Sons. (1979).

Litterst, C.L., Farber, T.M., Baker, A.M. and Van Loon, E.J. (1972). Effect of Polychlorinated Biphenyls on Hepatic Microsomal Enzymes in the Rat. Toxicology and Applied Pharmacology. 23:112-122.

Newell A.J., D.W. Johnson, and L.K. Allen. (1987). Niagara River Biota Contamination Project: Fish Flesh Criteria for Piscivorous Wildlife. NYS Department of Environmental Conservation Publication. (July 1987). Albany NY.

Norback D.H. and R.H. Weltman (1985). Polychlorinated Biphenyl Induction of Hepato-Cellular Carcinoma in the Sprague-Dawley Rat. Environ. Health Perspect. 60:97-105.

NYS Department of Environmental Conservation. Superfund and Contract Laboratory Protocol. (Jun. 1986).

O'Brien & Gere Engineers, Inc. Letter to John N. Hanson, Esq., Beveridge & Diamond. Diefenbach's Memo on Occupational Exposure to PCBs. (Apr. 22, 1985).

O'Brien & Gere Engineers, Inc. Scope of Work. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Jun. 1985).

O'Brien & Gere Engineers, Inc. Work Plan. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Jun. 1985).

O'Brien & Gere Engineers, Inc. Work Plan Supplement. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Dec. 1985).

O'Brien & Gere Engineers, Inc. Work Plan Supplement. Phase II. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Nov. 1986).

O'Brien & Gere Engineers, Inc. Quality Assurance Project Plan. Revision 4. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Nov. 1986).

O'Brien & Gere Engineers, Inc. Draft Site Investigation Report. Remedial Investigation/Feasibility Study. Crab Orchard National Wildlife Refuge. (Feb. 1987).

O'Connor, Harold J. (USFWS, Wash. DC). Memorandum to D. Rosenberger, U.S. FWS/RCA. Lead Contamination of White-Tailed Deer at the Crab Orchard National Wildlife Refuge, Illinois. (Jun. 22, 1984). 15

O'Toole, Michael (USEPA). Memorandum to Russell Diefenbach. Crab Orchard Creek National Wildlife Refuge, Marion, Illinois Trip Report. (Jul. 21, 1983). 52

O'Toole, Michael (USEPA, Chicago). Letter to D. Stalling, CNFRL. Transmittal of Fish & Sediment Analysis. (Jul. 17, 1984). 11

O'Toole, Michael (USEPA, Chicago). Memorandum to File. Sampling Effort at the Sangamo Dump on Aug. 15, 1984. (Sep. 17, 1984). 9

Paladino, Pete (IL DOC). Memorandum to Mike Carter. Contaminant Sampling of Crab Orchard Lake.

Palmer, E.L and Fowler, H.S. (1975). Fieldbook of Natural History, Second Ed., McGraw Hill Co., New York.

Paustenbach, D.J., Shu, H.P., and Murray, F.J. Critical Assumptions in Risk Assessment of Soil Contaminated with 2,3,7,8-TCDD (Dioxin). From Workshop on Assessing Risks from Chemical Contaminants in Soil. Andover, MA (May 8, 1986).

Porter, J. Winston (USEPA). Memorandum to EPA Regional Administrators, Regions I-X. Interim RCRA/CERCLA Guidance on Non-Contiguous Sites and On-Site Management of Waste and Treatment Residue. (Mar. 27, 1986).

Redmon, Walter L. (U.S. EPA Reg. V) Memorandum to Kenneth Fenner. Trip Report - Meeting at Crab Orchard National Wildlife Refuge on Contamination of the Refuge with PCBs, Lead and Other Chemicals. (Aug. 11, 1983). 47

Registry of Toxic Effects of Chemical Substances (RTEC). NIOSH. (June, 1983).

Ruelle, D. (1987). Telephone conversation between D. Ruelle, U.S. FWS Project Director, Rock Island, IL and H. Appleton, Senior Scientist, O'Brien & Gere, Syracuse, NY. December 10, 1987.

Ruelle, Richard (USFWS). Letter to C.B. Murphy, OBG Engineers. Comments to Draft RI Report. (Jun. 9, 1987).

Ruelle, Richard (FWS). Report to Refuge Manager. Mercury Levels in Crab Orchard Lake Largemouth Bass. (Feb. 1983). 38

Ruelle, Richard (FWS). Report to Refuge Manager. Survey for Lead on Crab Orchard National Wildlife Refuge. (Feb. 1983). 39

Ruelle, R. and Adams, R. (FWS) Survey for Polychlorinated Biphenyls In Some Abandoned Industrial Dumps and in Lake Sediments on Crab Orchard National Wildlife Regue. (Apr. 1984). 28

Ruelle, Richard (FWS). Memorandum to List. RI/FS Crab Orchard NWR. (Jul. 11, 1984). 13

Ruelle, Richard and Ross Adams (FWS). Survey for Metals in Deer Livers, and in Soils and Vegetation in Abandoned Industrial Dumps on Crab Orchard National Wildlife Refuge. (Jul. 1984).

Safe, S., Bandiera, S., Sawyer, T., Zmudzka, B., Mason, G., Romkes, M., Denomme, A., Sparling, J., Okey, A.B., Fujita, T. Effects of Structure on Binding to the 2,3,7,8-TCDD Receptor Protein and AHH Induction - Halogenated Biphenyls. Environ. Health Perspec. (1985). 61:21-34.

Sawyer, T. and Safe, S. PCB Isomers and Congeners: Induction of Aryl Hydrocarbon Hydroxylase and Ethoxyresorufin o-Deethylase Enzyme Activities in Rat Hepatoma Cells. Toxicol. (1982). Lett 13:87-94.

Sawyer, T.W., Safe, S. In Vitro AHH Induction by Polychlorinated Biphenyl and Dibenzofuran Mixtures: Additive Effects. Chemosphere. (1985). 14:79-84.

Sawyer T.W., Vatcher, A.D., Safe, S. Comparative Aryl Hydrocarbon Hydroxylase Induction Activities of Commercial PCBs in Wistar Rats and Rat Hepatoma H-4-II E cells in culture. Chemosphere. (1984). 13:695-701.

Schmidt, C.J. (1985). National Pesticide Monitoring Program. Arch. Environ. Contam. Toxicol., 14:225-60. As reported in ATSDR (1987).

Shawney, B.L and Hankin, L. (1984). Plant Contamination by PCBs from Amended Soils. J. Food Protection, V.47, N.3:232-236 (March, 1984).

Skea J.C., H.A. Simonin, E.J. Harris et al. 1979. Reducing Levels of Mirex, Aroclor 1254, and DDE by Trimming and Cooking Lake Ontario Brown Trout and Small Mouth Bass, Great Lakes Res. 5:153-159.

Stalling, D. L. Memorandum to ECL Specialist Region III. Dibenzofuran and Dioxin Residues in PCB contaminated Crab Orchard soil. (Sep. 9, 1983). 32

Stalling, David L. Memorandum to Dick Ruelle, Reg. III, RCA Specialist. Quality Control/Quality Assurance Statement for Crab Orchard RFP. (Sep. 24, 1984). 7

Suzuki, M., Aizawa, N., Okano, G. and Takahashi, T. (1977). Translocation of Polychlorinated Biphenyls in Soil into Plants: A Study by a Method of Culture of Soybean Sprouts. Arch. Environ. Contam. Toxicol. 5:343-352.

Thomas, Bob (DWPC). Memorandum to Jim Frank, DLPC. PCBs in Crab Orchard Lake/Sangamo Electric Company Inactive Dump Site. (May 4, 1984). 26

Updike, Gerald H. (FWS). Memorandum to John Ellis, Crab Orchard NWR. Hazardous Waste Site Cleanup Meeting Notes. (Jul. 2, 1984). 49

U.S.D.A. (United States Department of Agriculture). Big Muddy River Comprehensive Basin Study, Appendix K, Agriculture. (1968).

U.S.D.A. Soil Conservation Service. Williamson County Soils, Urbana, Illinois. (1959).

USEPA (1980). Ambient Water Quality Criteria for Polychlorinated Biphenyls. Washington, DC. (Oct., 1980). 881-117798.

USEPA (1984). Remedial Response Program. Hazardous Waste Site Listed Under CERCLA. (Jun. 15, 1984). 17

USEPA. (1984). Health Effects Assessment for Lead. ECAO-CIN-H055. Cincinatti, OH. (Sep., 1984).

USEPA (1985). Contract Laboratory Program Caucus Protocol - Multimedia, Multicomponent. (Jan., 1985).

USEPA (1985). Guidance on Remedial Investigations under CERCLA. Prepared for the Hazardous Waste Engineering Research Laboratory (Office of Research and Development) and Offices of Emergency and Remedial Response, and Waste Programs Enforcement (Office of Solid Waste and Emergency Response). (May, 1985).

USEPA (1985). Reference Values for Risk Assessment. First Draft. ECAO-CIN-477. Environmental Criteria and Assessment, Cincinnati, OH.

USEPA (1986). Superfund Exposure Assessment Manual, Draft. Office of Emergency and Remedial Response, U.S. Environmental Protection Agency. Washington DC. 20460. January, 1986.

USEPA/600/6-86/002 (May 1986). Office of Research and Development. U. S. Environmental Protection Agency. Washington DC 20460.

USEPA (1986). Development of Advisory Levels for PCBs Cleanup. (May, 1986)

USEPA (1986). Health and Environmental Effects Profile for Phthalic Esthers. ECAO-CIN-P188. (Sep., 1986).

USEPA (1986--) Health and Environmental Effects Profile on Nitrosamines. March 1986. ECAO. Cincinnati OH.

USEPA (1987). Memorandum from L.M. Thomas to Administrators and General Counsel. Interim Policy for Assessing Risks of Dioxins Other Than 2,3,7,8-TCDD. (Jan. 7, 1987).

USEPA (1987). Drinking Water Criteria Document for Polychlorinated Biphenyls (PCBs). Prepared by the Environmental Criteria and Assessment Office, U.S. EPA, Cincinnati, OH. May 1987. ECAO-CIN-414

USEPA (1987). OHEA Documentation of ADIs, qs, and Associated Relevant Data. Environmental Criteria and Assessment Office, Cincinnati, OH.

USEPA (1988). Risk Assessment for Dioxin Contamination, Midland, Michigan, Region V. Chicago, IL. EPA-905/4-88-005.

USEPA/Clement Associates, Inc. Chemical, Physical, and Biological Properties of Compounds present at Hazardous Waste Sites. See Clement Associates, Inc.

USEPA-DLPC, Illinois Memorandum. Proposed sampling of public water supplies. (May 31, 1984). 19

USEPA/ICF Inc.. Superfund Health Assessment Manual, Contract No. 68-01-6872. See ICF Incorporated.

Vleck, D. (1979). The Energy Cost of Burrowing by the Pocket Gopher (Thomomys bottae). Physiological Zoology V.52:122-136.

Williams, G.M., and Weisburger, J.H. Chemical Carcinogens. Casarett and Doull's Toxicology. 3rd ed. C.D. Klassen, M.O. Amden and J. Doull, eds. MacMillan Publishing Co. New York, N.Y. (1986).

Wolf, Greg (USFWS). Memorandum to Files. Historical Summary of Contaminants on CONWR. (May 31, 1984). 20

Woolf, Alan et al. Regional Variation in Metals in Livers of White-Tailed Deer in Illinois. Trans. Ill. State Academy of Science. (Jan., 1983). 1, 2:305-310. 40

Zabik M.E., P. Hoojjat, and C.M. Weaver, 1979. Polychlorinated Biphenyls, Dieldrin, and DDT in Lake Trout, Cooked by Broiling, Roasting, or Microwave. Bull, Environ. Contam. & Toxicol. 21:136-143.

Zabik, M.E., C. Merrill, and M.J. Zabik. 1982. PCBs and other Xenobiotics in Raw and Cooked Carp. Bull, Environ. Contam., Toxicol. 28:710-715.

REMEDIAL INVESTIGATION REPORT

CRAB ORCHARD NATIONAL WILDLIFE REFUGE

ATTACHMENT 1

MONITORING PLAN

TABLE OF CONTENTS

TABLE OF CONTENTS
(Continued)

SECTION 1 - INTRODUCTION

1.01 Purpose

This Monitoring Plan has been developed to provide the specifications for periodic routine monitoring surface waters at the Crab Orchard National Wildlife Refuge (CONWR). The purpose of this monitoring program is to ensure that water quality at CONWR does not deviate significantly from the conditions observed during the Remedial Investigation.

Quarterly reports presenting the results of the site monitoring will be prepared and filed at the Refuge headquarters. The quarterly reports are described in Section 3.05. Monitoring activities should continue for a period of thirty years or through the duration of active industrial operations at the Refuge.

1.02 General

The Crab Orchard National Wildlife Refuge is owned by the U.S. government and is currently administered by the U.S. Fish and Wildlife Service (FWS). During the past 40-50 years, several industrial tenants have leased land on the eastern portion of the refuge for manufacturing operations. Some industrial tenants have continued their operations on the eastern area, while the western portion of the refuge is a popular recreation area. Public access to the refuge is limited to authorized personnel on the eastern portion. A remedial investigation (RI) of 33 study sites at the refuge was completed in 1988 to evaluate the the existing conditions of Refuge sites, tributaries of Crab Orchard Lake and Crab Orchard Lake, a drinking water supply for the refuge. Six

sites were carried forward to a Feasibility Study evaluation of remedial alternatives due to the presence of some contaminants in soil or sediments. Eight sites (Table 1) were recommended for monitoring of surface waters due to their proximity to active industrial operations. Section 4 details the site specific monitoring locations and parameters for these eight monitoring sites.

SECTION 2 - SAMPLING PLAN

2.01 Sample Types

Surface water is the media of interest for the purposes of this Monitoring Plan. Waters from surface streams and Crab Orchard Lake will be sampled.

For the most part, all samples will be obtained as single grab samples. However, at some sites, composite samples will be prepared. The compositing procedure to be followed is discussed under the sampling procedures described below for each type of surface water that is to be sampled.

Field blanks, field duplicates and matrix spikes/matrix spike duplicates (MS/MSD) will also be collected or prepared as part of the quality assurance and quality control (QA/QC) requirements outlined in Section 3.

The number of samples to be collected for each of these categories will be dependent on the total number of samples to be collected. A description of each type of QA/QC sample follows. Field duplicates are two distinct samples taken from the same location at similar times using identical sampling equipment that has been decontaminated in a similar manner. One field duplicate will be collected for every ten samples collected. Field blanks for surface waters will be prepared using ultrapure distilled/dionized water. The field blank sample will be placed into the appropriate decontaminated sampling equipment, removed from the equipment and then placed into the proper sampling containers. One field blank will be collected for every twenty samples collected. Matrix spike (MS) samples are collected following the sampling

procedure for the matrix being investigated using the same procedure as for the field duplicate samples. Samples tagged as spikes will be treated with matrix spiking solutions in the laboratory and will be analyzed in duplicate (MS/MSD). One matrix spike sample will be collected for every twenty samples.

2.02 Sampling Procedures

2.02.01 Surface Water

Surface water samples should be taken from 2 to 5 (or more) points spaced equally across the width of the stream, seepage, or pond. The specific number of points may be determined in the field and should be adequate to accurately reflect the size of the body of water being sampled. At each point, subsamples should be collected, representative of the total depth of the body of water. The subsamples may then be composited into a single sample for analysis, dependent upon the intent of the sampling program. For small shallow streams, a single sample collected just below the surface at the stream's midpoint may be adequate for sampling and analyses purposes.

Whether samples are obtained from a boat, a bridge, or by wading into the water body, samples should be taken while facing upstream, away from the influent of the sampler or stream flow.

Collection is accomplished by submerging a precleaned, prelabeled container at the sampling point to the depth required. Sampling containers should be preserved with the appropriate preservatives as detailed in Table 2. Container types are also specified in Table 2 for the different analyses required. For deep

streams, or deep ponds, a Kemmerer, Van Dorn or other sampler specifically designed for this purpose may be used. For shallow (i.e. less than three feet deep) locations, an inverted sample container may be carefully submerged by hand and then slowly allowed to fill.

Samples should be stored in an insulated ice cooler at 4 degrees Celsius. All pertinent information should be recorded in the sampling log book and chain of custody forms, including sample collection date, location and identification.

2.02.02 Lake Water Columns

A composite column water sample from Crab Orchard Lake will be obtained as follows: discrete samples from the surface, mid-depth and approximately six inches from the bottom will be taken using a stainless steel Kemmerer, Van Dorn or other sampler specifically designed for this purpose. The sampling device will be decontaminated and rinsed with the water to be sampled prior to each sample collection.

Equal aliquots from each of the three depths at each site should be composited in the proper precleaned, prelabeled, preserved containers as necessary for the analyses to be completed. The samples should be stored in an insulated ice cooler at 4 degrees Celsius. All pertinent information should be recorded, including sample collection date, location and identification in the sampling log book and chain of custody forms.

2.03 Sample Control

2.03.01 Sample Containers and Preservation

Sample containers, sample preservation and filling instructions may be different for each type of analysis that is to be performed. Care must be taken to utilize the correct sample container(s) and preservative(s) to ensure the integrity of the samples. Table 2 provides a listing of the sample preservatives and the sample containers to be utilized. Sample containers will be supplied by the contract laboratory. The collected samples will be kept out of direct sunlight and, after decontamination and labeling, will be placed in coolers for shipment to the contract laboratory.

2.03.02 Sample Shipment and Chain of Custody

Samples will be packed and labelled according to DOT regulations and protocols. Samples will be shipped via a 24 hour delivery service to the contract laboratory so that the samples can be extracted within allowable time limits.

Chain of custody procedures must be followed closely to ensure that an accurate record of the collection, transport, analysis and disposal of the sample(s) is documented. The chain of custody procedures include field custody, laboratory custody and evidence files and conform with the procedures outlined in NEIC Policies and Procedures (EPA-3309-78-001-R).

Samples are accompanied by a field chain of custody record (Figure 1). When transferring the possession of samples, the individuals relinquishing and receiving will sign, date and note the time on the record. This procedure documents sample custody transfer. Samples will be packaged with a separate custody record

accompanying each shipment. All shipments will be accompanied by the field chain of custody record identifying its contents. The original record will accompany the shipment and a copy of the original record will be retained by the Project Coordinator.

Whenever samples are split with a source or government agency, it is noted in the "Remarks" section of the custody record. The note indicates with whom the samples are being split and is signed by both the sampler and recipient. If either party refuses a split sample, this will be noted and signed by both parties. The person relinquishing the samples to the facility or agency should request the signature of a representative of the appropriate party, acknowledging receipt of the samples. If a representative is unavailable or refuses to sign, this is noted in the "Remarks" space. When appropriate, as in the case where the representative is unavailable, the custody record should contain a statement that the samples were delivered to the designated location at the designated time.

If the samples are sent by mail, the package will be registered with return receipt requested. If sent by common carrier, a Government Bill of Lading will be used. Air freight shipments are sent collect. Freight bills, Post Office receipts and Bill of Lading will be retained as part of the permanent documentation.

All documents and raw data from the individual laboratories performing specific analysis will be transferred at the end of the monitoring period to the Refuge Manager for the Fish and Wildlife Service, CONWR for safekeeping for a period of ten years.

SECTION 3 - ANALYTICAL PROTOCOLS, QUALITY ASSURANCE AND CONTROLS

A brief discussion of analytical protocols, quality assurance and control procedures follows.

3.01 Calibration Procedures

3.01.01 Equipment

Generally, all field equipment will be calibrated in accordance with the manufacturer's instructions. Any field equipment that is not covered by the investigator's standard operating procedure will have a specific calibration and operation instruction sheet prepared for it.

3.01.02 Standards

Standards may be generally grouped into two classifications: primary and secondary. Primary standards include USP drugs, NBS and ASTM materials, and certain designated EPA reference material. All other standards are to be considered secondary. No testing of primary standards is necessary. Secondary standards will be examined when first received, and less stable standards will be rechecked at appropriate intervals, usually six months to one year.

3.01.03 Records

A records book will be kept for each standard. Each record will include name and date received, source, code or lot number,

purity, testing data, special storage requirements and storage location. Records will be kept on each instrument requiring calibration, to record all activities associated with maintenance, QA monitoring and repairs program.

3.02 Analytical Procedures

The analyses and methods detection limits for analytical parameters are given in Table 3. When analyzing samples by the listed standardized methods, the accuracy or precision of the data generated by the laboratory is determined through analyses of replicates, spiked samples, synthetic reference standard samples, and field and laboratory blanks along with each set of samples. Any interference is identified and documented. The required QA/QC samples to be collected are specified in Section 2.

3.03 Internal Quality Control

3.03.01 Analytical Procedures and Laboratory Quality Control

The quality control objectives for the monitoring analytical program are listed in Table 4. The frequency of replicate samples, spiked samples, reference samples and blanks, as well as control limits for acceptability are identified in this table.

Quality control data, which includes the analysis of EPA standard reference materials to verify initial calibration of non-CLP analysis, and reports of blanks, duplicates and spiked samples will be included with each package. The laboratory selected to perform the analytical procedures for this monitoring program will be certified by the appropriate State and Federal Agencies. The

laboratory should participate and meet acceptance criteria established by the Illinois inter-laboratory QA/QC programs for analyses of split samples and spiked standards.

3.03.02 Field Control and Preventive Maintenance

Field sampling crews will be under the direct supervision of a crew chief. Records will be used to document the collection of each sample.

Preventive maintenance procedures will be carried out on all field equipment in accordance with manufacturer's equipment manuals. Any field equipment that is not covered by the standard operating procedures will have a specific maintenance instruction sheet prepared for it.

3.04 Data Assessment and Validation

Data assessment will be based upon instrument tuning criteria, duplicate samples, surrogate recoveries, matrix spikes and matrix spike duplicates. Any data that should be rated as "unacceptable" or "preliminary" will be identified. Corrective actions will be identified if required.

Corrective action procedures are developed on a case- by-case basis. These actions may include:

- Reanalyzing samples if holding time requirements have not been exceeded.
- Altering field or handling procedures.
- Resampling.
- Using a different batch of sample containers.

- Recommending an audit of laboratory procedures.
- Accepting data with acknowledged level of uncertainty.
- Discard data.

3.05 Data Reporting

Each quarterly analytical data submission will contain QA/QC sections that summarize data quality information. The reports will include:

1. Discussion of accuracy, precision, completeness of data and results of performance and system audit specified in Table 4.
2. Discussion of results of data assessment.
3. Data results.
4. Chain of custody forms.

SECTION 4 - SITE SPECIFIC MONITORING PROGRAMS

4.01 Site 10 and Site 11 : Waterworks Drainage Channels

4.01.01 Site Descriptions

Sites 10 and 11 receive various drainage channels leading from active industrial operations within the Olin D and P Areas prior to their discharge to Crab Orchard Lake. These channels discharge near the Refuge Waterworks.

4.01.02 Baseline Parameters

Figure 2 depicts the surface water monitoring locations for Sites 10 and 11. The parameters selected for surface water monitoring at the sites include cyanide, iron, magnesium, manganese, mercury and phthalate esters. The results of the RI sampling at Sites 10 and 11 for the previously mentioned parameters are contained in Table 5 and will serve as baseline concentrations for the monitoring program at the sites.

4.01.03 Monitoring

Locations 10-1 and 10-3 at Site 10 and locations 11-1 and 11-3 at Site 11 will be the surface water monitoring locations for the two sites (Figure 2). Future monitoring at Sites 10 and 11 requires that composite surface water samples be collected quarterly from each location at each site. In addition, a full volatile and semi-volatile scan will be run annually on composite surface water samples collected from the four monitored locations at Sites 10 and 11. The samples should be collected and analyzed for the

parameters listed in Section 4.01.2 and in accordance with the procedures outlined in Sections 2 and 3.

4.02 Site 14 : Area 14 Solvent Storage Ditch

4.02.01 Site Description

Site 14 is a drainage ditch adjacent to the active manufacturing operations of Diagraph-Bradley. The ditch receives run-off from a manufacturing area where solvents are handled in bulk or in drums. The ditch runs north, parallel to the road that is west of the plant, and ultimately discharges to Crab Orchard Lake.

4.02.02 Baseline Parameters

Figure 3 and 4 depicts the surface water monitoring locations for Site 14. The parameters selected for surface water monitoring at the site include acetone, chloromethane, methylene chloride and phthalate esters. The results of the RI sampling at Site 14 for the previously mentioned parameters are contained in Table 6 and will serve as baseline concentrations for the monitoring program at the site.

4.02.03 Monitoring

Locations 14-1 and 14-4 at the Solvent Storage Area will be the surface water monitoring locations for Site 14 (Figure 3 and 4). Future monitoring at Site 14 requires that grab surface water samples be collected quarterly from each monitoring location at the site. The samples should be collected and analyzed for the

parameters listed in Section 4.02.02 and in accordance with the procedures outlined in Sections 2 and 3.

4.03 Site 16 : Area 7 Industrial Site

4.03.01 Site Description

Site 16 consists of a ditch within the Area 7 Industrial Site. The Area 7 Industrial Site is comprised of 33 buildings over an area of 55 acres which have been used for a variety of industrial purposes during the last forty years. Three of the buildings are used by Pennzoil in waste oil recovery and recycling operations and two other buildings are used by a refurbisher of mining equipment. The other buildings at the site are used for storage purposes or are abandoned. The Site 16 drainage ditch runs from south to north through the buildings and discharges to Crab Orchard Lake.

4.03.02 Baseline Parameters

Figure 5 depicts the surface water monitoring locations for Site 16. The parameters selected for monitoring at the site include chloromethane, carbon tetrachloride, aldrin, dieldrin and polynuclear aromatic hydrocarbons (PAH) in water. The results of the RI sampling at Site 16 for the previously mentioned parameters are contained in Table 7 and will serve as baseline concentrations for the monitoring program at the site.

4.03.03 Monitoring

Location 16-18 at the Area 7 Industrial Site will be the surface water monitoring location for the site (Figure 5). Future monitoring at Site 16 requires that composite surface water samples be collected quarterly from the single monitoring location at the site. This sample should be collected and analyzed for the parameters listed in Section 4.03.2 and in accordance with the procedures outlined in Sections 2 and 3.

4.04 Sites 25, 26, 27: Crab Orchard Creek

4.04.01 Site Descriptions

Site 25 consists of the Crab Orchard Creek upstream and downstream of the Marion Landfill as well as the adjacent pond. The Old Marion Landfill is located adjacent to Crab Orchard Creek on Old Creal Springs Road. This municipal landfill has been inactive for a number of years. A 3/4 acre pond is located next to the landfill.

Site 26 is located on Crab Orchard Creek downstream from the Marion Sewage Treatment Plant.

Site 27 is located on Crab Orchard Creek downstream of Interstate Route 57.

Sites 25 and 26 are not located within the boundaries of the Refuge and are not under the management of the U.S. FWS. Site 27 is located on the Refuge.

4.04.02 Baseline Parameters

Figure 6 identifies the surface water monitoring locations for Sites 25, 26 and 27. The parameters selected for surface water

monitoring at the sites include cyanide, magnesium, manganese, TOC and TOX in water. The results of the RI sampling at Sites 25, 26 and 27 for the previously mentioned parameters are contained in Table 8 and will serve as the baseline concentrations for the monitoring program at the sites.

4.04.03 Monitoring

Location 25-1, downstream of the municipal landfill location and location 27-1, below the I- 57 dredge area, will be the surface water monitoring locations for the sites (Figure 6). Future monitoring at Sites 25, 26 and 27 requires that composite surface water samples be collected quarterly from each monitoring location at each site. The samples should be collected and analyzed for the parameters listed in Section 4.04.2 and in accordance with the procedures outlined in Sections 2 and 3.

4.05 Site 34: Crab Orchard Lake

4.05.01 Site Description

Site 34 is comprised of Crab Orchard Lake. This lake was formed in 1940 by construction of a spillway across Crab Orchard Creek. The lake has a surface area of 6,965 acres, an average depth of 8-9 feet on the western portion and an average depth of two feet on the eastern portion. The retention time is 0.8 years, with a storage capacity of 72,525 acre-feet.

Water enters the lake through several creeks, including the Crab Orchard Creek at the eastern end of the lake. The eastern section of the lake has been bordered by manufacturing operations

since the 1940s. Water exits the lake through the spillway at the western end and through use of 280,000 gpd of lake water by the Refuge.

4.05.02 Baseline Parameters

Figure 7 identifies the surface water monitoring locations for Site 34. The parameters selected for monitoring at the site include arsenic and PCBs. The results of the RI sampling at Site 34 for the previously mentioned parameters are contained in Table 9 and will serve as the baseline concentrations for the monitoring program at the site.

4.05.03 Monitoring

Locations 34-6, 34-12, 34-14 and 34-15 will be the surface water monitoring locations for the site (Figure 7). Future monitoring at Site 34 require that composite water columns be collected quarterly from each location at the lake at three depths: the surface, mid-depth and approximately six inches from the bottom. The sample should be collected and analyzed for the parameters listed in Section 4.04.2 and in accordance with the procedures detailed in Sections 2 and 3.

TABLES

TABLE 1

MONITORING PLAN
STUDY SITES

SITES 10 and 11 : WATERWORKS DRAINAGE CHANNELS

SITE 14 : AREA 14 SOLVENT STORAGE DITCH

SITE 16 : AREA 7 INDUSTRIAL SITE DITCH

SITES 25, 26 AND 27 : CRAB ORCHARD CREEK

SITE 34 : CRAB ORCHARD LAKE

TABLE 2

SAMPLE PRESERVATION AND CONTAINERS

PARAMETER	WATER
Volatiles	40 ml vial (2)/4 deg.C
Base/Neutrals/Acids	1 liter glass
Pesticide/PCB	1 quart glass (teflon)
PCBs Low Level (water)	1 quart glass (teflon)
Metals	1 pint plastic/HNO3
Cyanide	1 pint plastic/NaOH
Indicators - pH	1 pint plastic

TABLE 3 ATTACHMENT

PROCEDURES FOR LOW-LEVEL PCBs IN WATER

5.0 EXTRACTION PROCEDURE FOR LOW LEVEL PCBs IN WATER

5.1 Determine pH of sample and adjust to a range of 5-9 with .1:1 sulfuric acid solution or in sodium hydroxide. Transfer the entire contents of sample bottle into a 2 liter separatory funnel.

5.2 Add 1ml of 50ppt DBC or equivalent (compound which will not be removed by cleanup options) surrogate solution to the sample.

5.3 To every batch of samples for low level analysis, add a blank, a matrix spike (MS) and a matrix spike duplicate (MSD). The blank is 2000ml of organic-free water and it is treated in the same manner as the samples. The MS and MSD samples are spiked in duplicate. Ideally, a sample is collected in triplicate in the field in 3 separate containers. The first sample is the sample itself. The second and third samples are spiked with the following compound in acetone:

Note: See Tables 4 a 5 for detection level and QA/QC requirement

Aroclor 1254	5ppt

5.4 Add 50ml of 15% methylene chloride in hexane to the sample jar, seal and shake. Transfer bottle extract to separatory funnel and extract the sample by shaking vigorously for 2 minutes with periodic venting to release pressure.

5.5 Drain water sample back into sample jar. Drain the hexane extract through a sodium sulfate funnel into a Kuderna-Danish evaporator. Return water sample back to separatory funnel and repeat extraction 2 more times combining all extracts into the Kuderna-Danish evaporator.

5.6 If emulsion problems occur during extraction, collect all 3 extracts in a 200ml centrifuge bottle without sodium sulfate drying. Centrifuge the contents of the bottle until 2 distinct layers are formed. Transfer the top layer (the hexane extract) through sodium sulfate into the Kuderna-Danish evaporator.

5.7 Concentrate extract to 0.2ml.

5.8 If cleanup is necessary, refer to Section 7.0

5.9 Concentrate final extract after cleanup to 0.2ml and analyze by GC/ECD.

LOW LEVEL PCBs

EXTRACTION PROCEDURE

FOR WATER MATRICES

August 1986

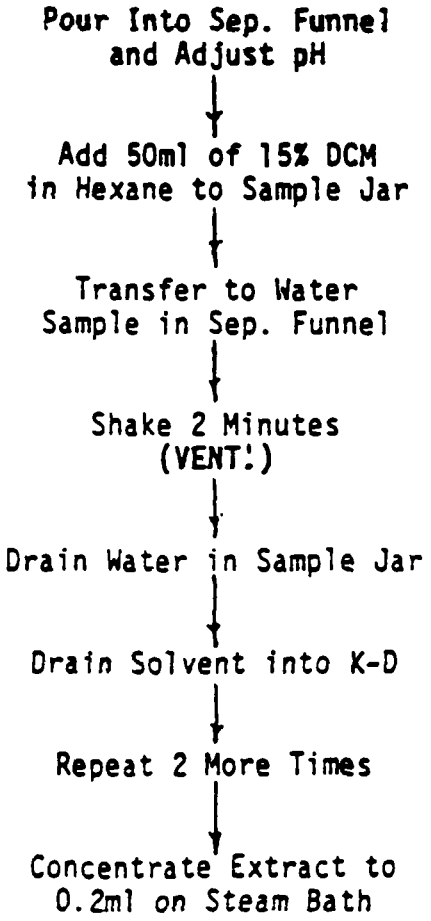


TABLE 4

QUALITY ASSURANCE REQUIREMENTS

ORGANICS

AUDIT	FREQUENCY	CONTROL LIMITS
Reagent Blank	1 per case or 10% of sample shipment.	Less than CRQL.
Field Blank	5% of sample shipment provided by sampling crew.	Less than CRQL.
Matrix Spike	10% of similar concentration/ matrix.	As specified by 40 CFR 136.
Matrix Spike Duplicate	10% of similar concentration/ matrix.	Recoveries as specified in method.

INORGANICS

AUDIT	FREQUENCY	CONTROL LIMITS
Reagent Blank	1 per case or 10% of sample shipment.	Less than CRQL.
Field Blank	5% of sample shipment provided by sampling crew.	Less than CRQL.
Field Duplicate	10% of sample shipment provided by sampling crew.	RPD within 20%.
Matrix Spike	10% of similar concentration/ matrix.	Recoveries as specified in method.

CRQL = Contract Required Quantitation Limit

Table 3
Page 1 of 5

ANALYTICAL METHODS
VOLATILES

PARAMETER	METHOD (1)	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
acetone	602	5	See Table 4
carbon tetrachloride	601	10	"
chloromethane	601	10	"
methylene chloride	601	10	"

(1) Code of Federal Regulations, Title 40, Section 136, Appendix A, "Test Procedure for Analysis of Organic Pollutants."

Table 3
Page 2 of 5

ANALYTICAL METHODS
SEMI-VOLATILES

PARAMETER	METHOD (1)	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
acenaphthene	610	5	See Table 4
acenaphthylene	610	5	"
anthracene	610	5	"
benzo(a)anthracene	610	5	"
benzo(b)fluoranthene	610	5	"
benzo(k)fluoranthene	610	5	"
benzo(a)pyrene	610	5	"
benzo(g,h,i)perylene	610	5	"
bis (2-ethylhexyl) phthalate	610	5	"
butyl benzyl phthalate	610	5	"
chrysene	610	5	"
dibenzo(a,h)anthracene	610	5	"
diethylphthalate	610	5	"
dimethyl phthalate	610	5	"
di-n-butyl phthalate	610	5	"
di-n-octylphthalate	610	5	"
fluorene	610	5	"
fluoranthene	610	5	"
indeno(1,2,3-cd)pyrene	610	5	"
naphthalene	610	5	"
phenanthrene	610	5	"
pyrene	610	5	"

(1) Code of Federal Regulations, Title 40, Section 136, Appendix A, "Test Procedure for Analysis of Organic Pollutants."

ANALYTICAL METHODS
PESTICIDES/PCBS

PARAMETER	METHOD (1)	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
aldrin	608	50	See Table 4
dieldrin	608	50	"
arochlor-1016	*	0.005	"
arochlor-1221	*	0.005	"
arochlor-1232	*	0.005	"
arochlor-1242	*	0.005	"
arochlor-1248	*	0.005	"
arochlor-1254	*	0.005	"
arochlor-1260	*	0.005	"

(1) Code of Federal Regulations, Title 40, Section 136, Appendix A, "Test Procedure for Analysis of Organic Pollutants."

* See Attachment 1

ANALYTICAL METHODS
METALS AND CYANIDE

PARAMETER	METHOD (2)	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
arsenic -T	206.2	10	See Table 4
iron -T	236.1	10	"
magnesium -T	242.1	10	"
manganese -T	243.1	10	"
mercury -T	245.1	0.5	"
cyanide	335.2	50	"

(2) Methods for Chemical Analysis of Water and Wastes, EPA-600/4-79-020

ANALYTICAL METHODS
INDICATORS

PARAMETER	METHOD	CONTRACT REQUIRED QUANTITATION LIMIT (ppb)	QA/QC CONTROLS
total organic carbon	415.1 (2)	1000	See Table 4
total organic halides	450.1 (3)	10	See Table 4

(2) Methods for Chemical Analysis of Water and Wastes, EPA-600/4-79-020

(3) EPA Method Study 32, EPA/600/54-55/080 (NTIS: PB86136538/AS)

TABLE 5

SITE BASELINE CONCENTRATIONS
SITE 10 AND SITE 11
WATERWORKS DRAINAGE CHANNELS

Baseline Concentrations In Surface Waters

		Site 10		Site 11	
PARAMETER	UNITS	10-1	10-3	11-1	11-3
Cyanide	ug/l	<0.05	<0.05	<0.05	<0.05
Iron (total)	ug/l	600	-	<300	-
Magnesium (total)	mg/l	9.85	-	10.6	-
Manganese (total)	mg/l	0.27	-	0.095	-
Mercury	ug/l	<0.5	-	<0.5	<0.6
Bis (Ethylhexyl) Phthalate	ug/l	-	<20 DR	-	-
Butyl Benzyl Phthalate	ug/l	-	<20 DR	-	-
Di-N-Butyl Phthalate	ug/l	-	<20 DR	-	-
Di-N-Octyl Phthalate	ug/l	-	<20 DR	-	-
Diethyl Phthalate	ug/l	-	<20 DR	-	-
Dimethyl Phthalate	ug/l	-	<20 DR	-	-
Acid Extractable Screen	ug/l	<100	-	<100	-
Base Neutral Screen	ug/l	<100	-	<100	-

KEY:
(-) = Analysis Not Run.
D = Holding time exceeded for extraction.
R = MS/MSD % recovery outside of QC limits.

TABLE 6

SITE BASELINE CONCENTRATIONS
SITE 14
AREA 7 SOLVENT STORAGE DITCH

Baseline Concentrations In Surface Waters

PARAMETER	UNITS	Locations 14-1	14-3	14-5	14-5 (Dup)
Methylene Chloride	ug/l	<1	<1	4 BI	15 BI
Acetone	ug/l	-	-	43 BI	36 BI
Chloromethane .	ug/l	<1	<1	<4 I	<4 I
Bis (Ethylhexyl) Phthalate	ug/l	-	-	<49 R	<20 R
Butyl Benzyl Phthalate	ug/l	-	-	<49 R	<20 R
Di-N-Butyl Phthalate	ug/l	-	-	<49 R	<20 R
Di-N-Octyl Phthalate	ug/l	-	-	<49 R	<20 R
Diethyl Phthalate	ug/l	-	-	<49 R	<20 R
Dimethyl Phthalate	ug/l	-	-	<49 R	<20 R
Acid Extractable Screen	ug/l	<100	<100	-	-
Base Neutral Screen	ug/l	<100	<100	-	-

KEY:
(-) = Analysis Not Run.
B = Indicates that the analyte was found in the blank; the level found in the sample was less than 10 times the level found in the blank or or, additionally in the case of VOAs the level found in the sample was less than 60 ppb of methylene chloride or acetone, which were found in the blank.
I = MS/MSD RPD outside QC limits.
R = MS/MSD % recovery outside of QC limits.

TABLE 8

SITE BASELINE CONCENTRATIONS
SITES 25, 26 and 27
CRAB ORCHARD CREEK

Baseline Concentrations In Surface Waters

		Locations					
PARAMETER	UNITS	25-3 Upstream of Marion Landfill	25-5 Pond Adjacent to Marion Landfill	25-1 Downstream of Marion Landfill	26-3 Upstream of Marion STP	26-1 Downstream of Marion STP	27-1 Below I-57 Dredge Area
Cyanide	mg/l	<0.05 C	-	<0.05 C	<0.05 C	<0.05 C	<0.05 C
Magnesium (total)	mg/l	14.3	5.6	47.2	18.2	26.5	12.4
Manganese (total)	mg/l	0.68	0.72	1.5	0.745	0.30	0.64
TOC	mg/l	11.0	18.0	7.0	0.6	5.0	8.0
TOX	mg/l	0.018	0.011	0.012	0.049	0.120	0.043

KEY:
(-) = Analysis not run.
C = Usable for qualitative interpretation only.

TABLE 7

SITE BASELINE CONCENTRATIONS
SITE 16
AREA 7 INDUSTRIAL SITE

Baseline Concentrations in Surface Waters

PARAMETER	UNITS	Locations 16-1	16-3	16-18
Chloromethane	ug/l	<1	<1	<4 A
Carbon Tetrachloride	ug/l	<1	<1	<4 A
Aldrin	ug/l	<10	<10	0.17 RIN
Dieldrin	ug/l	<10	<10	0.54 RIN
Acenaphthylene	ug/l	-	-	<38 DI
Acenaphthene	ug/l	-	-	<38 DI
Anthracene	ug/l	-	-	<38 DI
Benzo(a)anthracene	ug/l	-	-	<38 DI
Benzo(a)pyrene	ug/l	-	-	<38 DI
Benzo(b)fluoranthene	ug/l	-	-	<38 DI
Benzo(ghi)perylene	ug/l	-	-	<38 DI
Benzo(k)fluoranthene	ug/l	-	-	<38 DI
Chrysene	ug/l	-	-	<38 DI
Dibenzo(a,h)anthracene	ug/l	-	-	<38 DI
Fluoranthene	ug/l	-	-	<38 DI
Fluorene	ug/l	-	-	<38 DI
Indeno(1,2,3-cd)pyrene	ug/l	-	-	<38 DI
Naphthalene	ug/l	-	-	<38 DI
Phenanthrene	ug/l	-	-	<38 DI
Pyrene	ug/l	-	-	<38 DI
Acid Extractable Screen	ug/l	<100	<100	-
Base Neutral Screen	ug/l	<100	<100	-

Key:
(-) = Analysis Not Run.
A = Holding time exceeded for analysis.
D = Holding time exceeded for extraction.
I = MS/MSD RPD outside QC limits.
N = % breakdown level of DDT/Endrin exceeded in previous Eval Mix B-Pest/PCB
R = MS/MSD % recovery outside of QC limits.

TABLE 9

SITE BASELINE CONCENTRATIONS
SITE 34
CRAB ORCHARD LAKE

Baseline Concentrations In Water Columns

Locations

PARAMETER	UNITS	34-6 1(B)	34-7 2(C)	34-8 3(G)	34-9 4(N)	34-10 5(A)	34-11 6(D)	34-12 7(E)	34-13 8(F)	34-14 9(I)	34-15 10(J)
Arsenic (total)	ug/l	3.4 M	2.7	3.2 M	<2.5 M	<2.5 M	<2.5 M	2.00	2.00	<2.5 M	<2.00
PCBs	ug/l	0.008	0.019	<0.005	<0.005	<0.005	0.009	<0.005	<0.005	<0.005	<0.005

KEY:
(-) = Analysis not run.
M = Values determined by a one point standard addition.

LEAD

Summary

Lead is a heavy metal that exists in one of three oxidation states, 0, +2, and +4. There is suggestive evidence that some lead salts are carcinogenic inducing kidney tumors in mice and rats. Lead is also a reproductive hazard, and it can adversely affect the brain and central nervous sytem by causing encephalopathy and peripheral neuropathy. Chronic exposure to low levels of lead can cause subtle learning disabilities in children. Exposure to lead can also cause kidney damage and anemia, and it may have adverse effects on the immune system. The EPA Ambient Water Quality Criterion for the protection of human health and the Interim Primary Drinking Water Standard are both 50 µg/liter.

CAS Number: 7439-92-1

Chemical Formula: Pb

IUPAC Name: Lead

Chemical and Physical Properties

Atomic Weight: 207.19

Boiling Point: 1,740°C

Melting Point: 327.502°C

Specific Gravity: 11.35 at 20°C

Solubility in Water: Insoluble; some organic compounds are soluble

Solubility in Organics: Soluble in HNO_3 and hot concentrated H_2SO_4

Transport and Fate

Some industrially produced lead compounds are readily soluble in water. However, metallic lead and the common lead minerals are insoluble in water. Natural compounds of lead are not usually mobile in normal surface or groundwater because the lead leached from ores becomes adsorbed by ferric hydroxide or combines with carbonate or sulfate ions to form insoluble compounds.

Movement of lead and its inorganic and organolead compounds as particulates in the atmosphere is a major environmental transport process. Lead carried in the atmosphere can be removed by either wet or dry deposition. Although little evidence is available concerning the photolysis of lead compounds in natural waters, photolysis in the atmosphere occurs readily. These atmospheric processes are important in determining the form of lead entering aquatic and terrestrial systems.

The transport of lead in the aquatic environment is influenced by the speciation of the ion. Lead exists mainly as the divalent cation in most unpolluted waters and becomes adsorbed into particulate phases. However, in polluted waters organic complexation is most important. Volatilization of lead compounds probably is not important in most aquatic environments.

Sorption processes appear to exert a dominant effect on the distribution of lead in the environment. Adsorption to inorganic solids, organic materials, and hydrous iron and manganese oxides usually controls the mobility of lead and results in a strong partitioning of lead to the bed sediments in aquatic systems. The sorption mechanism most important in a particular system varies with geological setting, pH, Eh, availability of ligands, dissolved and particulate ion concentrations, salinity, and chemical composition. The equilibrium solubility of lead with carbonate, sulfate, and sulfide is low. Over most of the normal pH range, $PbCO_3$, and $PbSO_4$ control solubility of lead in aerobic conditions, and PbS and Pb control solubility in anaerobic conditions. Lead is strongly complexed to organic materials present in aquatic systems and soil. Lead in soil is not easily taken up by plants, and therefore its availability to terrestrial organisms is somewhat limited.

Bioaccumulation of lead has been demonstrated for a variety of organisms, and bioconcentration factors are within the range of 100-1,000. Microcosm studies indicate that lead is not biomagnified through the food chain. Biomethylation of lead by microorganisms can remobilize lead to the environment. The ultimate sink of lead is probably the deep oceans.

Health Effects

There is evidence that several lead salts are carcinogenic in mice or rats, causing tumors of the kidneys, as a result of either oral or parenteral administration. Data concerning the carcinogenicity of lead in humans are inconclusive. Available data are not sufficient to evaluate the carcinogenicity of organic lead compounds or metallic lead. There is equivocal evidence that exposure to lead causes genotoxicity in humans and animals. Available evidence indicates that lead presents a hazard to reproduction and exerts a toxic effect on conception, pregnancy, and the fetus in humans and experimental animals.

Many lead compounds are sufficiently soluble in body fluids to be toxic. Exposure of humans or experimental animals to lead can result in toxic effects in the brain and central nervous system, the peripheral nervous system, the kidneys, and the hematopoietic system. Chronic exposure to inorganic lead by ingestion or inhalation can cause lead encephalopathy, and severe cases can result in permanent brain damage. Lead poisoning may cause peripheral neuropathy in adults and children, and clinically undetectable permanent-learning disabilities in chil- dren may be caused by exposure to relatively low levels. Short-term exposure to lead can cause reversible kidney damage, but prolonged exposure at high concentrations may result in pro- gressive kidney damage and possibly kidney failure. Anemia, due to inhibition of hemoblobin synthesis and a reduction in the life- span of circulating red blood cells, is an early manifestation of lead poisoning. Several studies with experimental animals suggest that lead may interfere with various aspects of the immune response.

Toxicity to Wildlife and Domestic Animals

Freshwater vertebrates and invertebrates are more sensitive to lead in soft water than in hard water. At a hardness of about 50 mg/liter $CaCO_3$, the median effect concentrations for nine families range from 140 µg/liter to 236,600 µg/liter. Chronic values for Daphnia magna and the rainbow trout are 12.26 µg/liter and 83.08 µg/liter, respectively, at a hardness of about 50 mg/liter. Acute-chronic ratios calculated for three freshwater species ranged from 18 to 62. Bioconcentration factors, ranging from 42 for young brook trout to 1,700 for a snail, were reported. Freshwater algae show an inhibition of growth at concentrations above 500 µg/liter.

Acute values for twelve saltwater species range from 476 µg/liter for the common mussel to 27,000 µg/liter for the soft-shell clam. Chronic exposure to lead causes adverse effects in mysid shrimp at 37 µg/liter, but not at 17 µg/liter. The acute-chronic ratio for this species is 118. Reported bioconcentration factors range from 17.5 for the Quahog clam to 2,570 for the blue mussel. Saltwater algae are adversely affected at approximate lead concentrations as low as 15.8 µg/liter.

Although lead is known to occur in the tissue of many free-living wild animals, including birds, mammals, fishes, and invertebrates, reports of poisoning usually involve waterfowl. There is evidence that lead, at concentrations occasionally found near roadsides and smelters, can eliminate or reduce populations of bacteria and fungi on leaf surfaces and in soil. Many of these microorganism play key roles in the decomposer food chain.

Cases of lead poisoning have been reported for a variety of domestic animals, including cattle, horses, dogs, and cats. Several types of anthropogenic sources are cited as the source of lead in these reports. Because of their curiosity and their indiscriminate eating habits, cattle experience the greatest incidence of lead toxicity among domestic animals.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed Criteria)

The concentrations below are for active lead, which is defined as the lead that passes through a 0.45-μm membrane filter after the sample is acidified to pH 4 with nitric acid.

Freshwater

Acute toxicity: $e^{(1.34[\ln(\text{hardness})] - 2.014)}$ μg/liter

Chronic toxicity: $e^{(1.34[\ln(\text{hardness})] - 5.245)}$ μg/liter

Saltwater

Acute toxicity: 220 μg/liter
Chronic toxicity: 8.6 μg/liter

Human Health

Criterion: 50 μg/liter

Primary Drinking Water Standard: 50 μg/liter

NIOSH Recommended Standard: 0.10 mg/m^3 TWA (inorganic lead)

OSHA Standard: 50 $\mu g/m^3$ TWA

ACGIH Threshold Limit Values:

0.15 mg/m^3 TWA (inorganic dusts and fumes)
0.45 mg/m^3 STEL (inorganic dusts and fumes)

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

DOULL, J., KLAASSEN, L.D., and AMDUR, M.O., eds. 1980. Casarett and Doull's Toxicology: The Basic Science of Poisons. 2nd ed. Macmillan Publishing Co., New York. 778 pages

INTERNATIONAL AGENCY FOR RESEARCH ON CANCER (IARC). 1980. IARC Monographs on the Evaluation of Carcinogenic Risk of Chemicals to Humans. Vol. 23: Some Metals and Metallic Compounds. World Health Organization, Lyon, France. Pp. 325-415

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

NRIAGU, J.O., ed. 1978. The Biogeochemistry of Lead in the Environment: Part B. Biological Effects. Elsevier/North-Holland Biomedical Press, New York. 397 pages

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1977. Air Quality Criteria for Lead. Office of Research and Development, Washington, D.C. December 1977. EPA-600/8-77-017

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Lead. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-057

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1983. Draft Revised Section B of Ambient Water Quality Criteria for Lead. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. August 1983

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Lead. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H055 (Final Draft)

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2,332 pages

WORLD HEALTH ORGANIZATION. 1977. Environmental Health Criteria: 3. Lead. World Health Organization, Geneva. 160 pages

Summary

Copper is among the more mobile metals in the environment. It is toxic to humans at high levels; it causes irritation following acute exposure and anemia following chronic exposure. Sheep are very susceptible to copper toxicosis as are many aquatic organisms. The EPA Ambient Water Quality Criterion for copper is 1 mg/liter based solely on its organoleptic properti

Background Information

Copper exists in a valence state of +1 or +2. It is a lustrous, reddish metal. The physical properties of copper include ductility and conductivity of heat and electricity. Copper is found in nature as sulfide, oxide, or carbonate ore.

CAS Number: 7440-50-8

Chemical Formula: Cu

IUPAC Name: Copper

Chemical and Physical Properties

Atomic Weight: 63.546

Boiling Point: 2567°C

Melting Point: 1083°C

Specific Gravity: 8.92

Solubility in Water: Most copper salts are insoluble with the exception of $CuSO_4$, $Cu(NO_3)_2$, and $CuCl_2$ (the more common copper salts). The metal is insoluble in water.

Vapor Pressure: 1 mm Hg at 1628°C

Transport and Fate

Copper has two oxidation states, +1 (cuprous) and +2 (cupric Cuprous copper is unstable in aerated water over the pH range of most natural waters (6 to 8) and oxidizes to the cupric state. Several processes determine the fate of copper in the

humic substances; sorption to hydrous metal oxides, clays, and organic materials; and bioaccumulation. In waters polluted with soluble organic material, complexation with organic ligands can occur, thus favoring the prolonged dispersion of copper in solution. The presence of organic acids also can lead to the mobilization of copper from the sediments to solution. Copper has a strong affinity for hydrous iron and manganese oxides, clays, carbonate minerals, and organic matter. Sorption to these materials, both suspended in the water column and in the sediment, results in relative enrichment of the solid phase and reduction in dissolved levels. Sorption processes are quite efficient in scavenging dissolved copper and in controlling its mobility in natural unpolluted streams. The amounts of the various copper compounds and complexes that actually exist in solution depend on the pH, temperature, alkalinity, and concentrations of other chemical species. The levels of copper able to remain in solution are directly dependent on water chemistry. Generally, ionic copper is more soluble in low pH waters and less soluble in high pH waters.

As an essential nutrient, copper is accumulated by plants and animals, although apparently it is not generally biomagnified. Since copper is strongly bioaccumulated, and because biogenic ligands play an important role in complexing copper, biological activity is a major factor in determining the distribution and occurrence of copper in the ecosystem. For example, bioaccumulation patterns may exhibit seasonal variations related to biological activity.

Because many copper compounds and complexes are readily soluble, copper is among the more mobile heavy metals in soil and other surface environments. The major process that limits the environmental mobility of copper is adsorption to organic matter, clays, and other materials. Atmospheric transport of copper compounds can also occur.

Health Effects

Copper appears to increase the mutagenic activity of triose reductone and ascorbic acid in bacterial test systems. However, copper itself does not appear to have mutagenic effects in animals or humans. Copper does not appear to produce teratogenic or carcinogenic effects in animals or humans. Dietary levels of trace elements such as molybdenum, sulphur, zinc, and iron can affect the level of copper that produces certain deficiency or toxicity symptoms. In general, more attention is given to the problems associated with copper deficiency than to problems of excess copper in the environment. However, high levels of copper can be toxic to humans.

Metallic copper dust exposure can cause a short-term illness similar to metal fume fever that is characterized by chills, fever, aching muscles, dryness of mouth and throat, and headache. Exposure to copper fumes can produce upper respiratory tract irritation, a metallic or sweet taste, nausea, metal fume fever, and sometimes discoloration of skin and hair. Individuals exposed to dusts and mists of copper salts may exhibit congestion of nasal mucous membranes, sometimes of the pharynx, and occassionally ulceration with perforation of the nasal septum.

If sufficent concentrations of copper salts reach the gastrointestinal tract, they act as irritants and can produce salivation, nausea, vomiting, gastritis, and diarrhea. Elimination of ingested ionic copper by vomiting and diarrhea generally protects the patient from more serious systemic toxic effects which can include hemolysis, hepatic necrosis, gastrointestinal bleeding, oliguria, azotemia, hemoglobinuria, hematuria, proteinuria, hypotension, tachycardia, convulsions, and death. Chronic exposure may result in anemia.

Copper salts act as skin irritants producing an itching eczema. Conjunctivitis or even ulceration and turbidity of the cornea may result from direct contact of ionic copper with the eye.

Toxicity to Wildlife and Domestic Animals

Mean acute toxicity values for a large number of freshwater animals range from 7.2 µg/liter for Daphnia pulicaria to 10,200 µg/liter for the bluegill. Toxicity tends to decrease as hardness, alkalinity, and total organic carbon increase. Chronic values for a variety of freshwater species range from 3.9 µg/liter for brook trout to 60.4 µg/liter for northern pike. Hardness does not appear to affect chronic toxicity. The acute-chronic ratios for different species range from 3 to 156. The more sensitive species tend to have lower ratios than the less sensitive species. In addition, the ratio seems to increase with hardness. Acute toxicity values for saltwater organisms range from 17 µg/liter for a calanoid copepod to 600 µg/liter for the shore crab. A chronic value of 54 µg/liter and an acute-chronic ratio of 3.4 is reported for the mysid shrimp. Long term exposure to 5 µg/liter is fatal to the bay scallop.

Bioconcentration factors in freshwater species range from zero for the bluegill to 2,000 for the alga Chlorella regularis. Among saltwater species the highest bioaccumulation factors are those for the bivalve molluscs. Oysters can bioaccumulate copper up to 28,200 times without any significant mortality.

Sheep are very susceptible to copper toxicosis and poisoning may be acute or chronic. Acute poisoning is caused by direct

action of copper salts on the gastrointestinal [illegible] in gastroenteritis, shock, and death. The toxic dose is about 200 mg/kg and is usually obtained through an accidental overdose of an antihelminthic. Ingestion of excess copper over a long period of time results in absorption and accumulation of copper by the liver. This type of chronic cumulative poisoning may suddenly develop into an acute hemolytic crisis. Copper intake of 1.5 g/day for 30 days is known to be fatal for many breeds of sheep. Excessive copper may be stored in the liver as a result of excess copper ingestion, as a consequence of impaired liver function, or in connection with a deficiency or excess of other trace elements. Sheep eliminate accumulated copper very slowly after cessation of exposure.

Swine develop copper poisoning at levels of 250 mg/kg in the diet unless zinc and iron levels are increased. Toxicosis develops with hypochromic microcytic anemia, jaundice, and marked increases in liver and serum copper levels as well as serum aspartate amino transferase. High copper levels may be found in swine because of the practice of feeding high copper diets in order to increase daily weight gain. However, swine rapidly eliminate copper once it is removed from the diet. Cattle are much more resistant to copper in the diet than sheep or swine. Copper toxicity in ruminants can be counteracted by inclusion of molybdenum and sulfate in the diet.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed)

Freshwater

Acute toxicity: $e^{(0.905\ [\ln(\text{hardness})]\ -\ 1.413)}$ µg/liter

Chronic toxicity: $e^{(0.905\ [\ln(\text{hardness})]\ -\ 1.785)}$ µg/liter

Saltwater

Acute toxicity: 3.2 µg/liter
Chronic toxicity: 2.0 µg/liter

Human Health

Organoleptic criterion: 1 mg/liter

National Secondary Drinking Water Standards (USEPA): 1 mg/liter

OSHA Standards: 1.0 mg/m^3 TWA (dust and mist)
0.1 mg/m^3 TWA (fume)

ACGIH Threshold Limit Values: 1.0 mg/m^3 TWA (dusts and mists)
0.2 mg/m^3 TWA (fume)
2.0 mg/m^3 STEL (dusts and mists)

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

BOSTWICK, J.L. 1982. Copper toxicosis in sheep. J. Am. Vet. Med. Assoc. 180:386-387

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

UNDERWOOD, E.J. 1979. Trace metals in humans and animal health. J. Hum. Nutr. 35:37-48

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Copper. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-036

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Water quality criteria: Request for comments. Fed. Reg. 49:4551-4553

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Copper. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H025 (Final Draft)

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2,332 pages

CADMIUM

Summary

Cadmium is a metal that can be present in a variety of chemical forms in wastes or in the environment. Some forms are insoluble in water, but cadmium is relatively mobile in the aquatic environment. Cadmium is carcinogenic in animals exposed by inhalation and may also be in humans. It is uncertain whether it is carcinogenic in animals or humans exposed via ingestion. Cadmium is a known animal teratogen and reproductive toxin. It has chronic effects on the kidney, and background levels of human exposure are thought to provide only a relatively small margin of safety for these effects. The EPA Ambient Water Quality Criterion for protection of human health is 10 µg/liter.

Background Information

Cadmium is a soft, bluish white metal that is obtained as a by-product from the treatment of the ores of copper, lead, and iron. Cadmium has a valence of +2 and has properties similar to those of zinc. Cadmium forms both organic and inorganic compounds. Cadmium sulfate is the most common salt.

CAS Number: 7440-43-9

Chemical Formula: Cd

IUPAC Name: Cadmium

Chemical and Physical Properties

Atomic Weight: 112.41

Boiling Point: 765°C

Melting Point: 321°C

Specific Gravity: 8.642

Solubility in Water: Salts are water soluble; metal is insoluble

Solubility in Organics: Variable, based on compound

Vapor Pressure: 1 mm Hg at 394°C

Transport and Fate

Cadmium is relatively mobile in the aquatic environment compared to other heavy metals. It is removed from aqueous media by complexing with organic materials and subsequently being adsorbed to the sediment. It appears that cadmium moves slowly through soil, but only limited information on soil transport is available. Cadmium uptake by plants is not a significant mechanism for depletion of soil accumulations but may be significant for human exposure.

Health Effects

There is suggestive evidence linking cadmium with cancer of the prostate in humans. In animal studies, exposure to cadmium by inhalation caused lung tumors in rats, and exposure by injection produced injection-site sarcomas and/or Leydig-cell tumors. An increased incidence of tumors has not been seen in animals exposed to cadmium orally, but four of the five available studies were inadequate by current standards.

The evidence from a large number of studies on the mutagenicity of cadmium is equivocal, and it has been hypothesized that cadmium is not directly mutagenic but impedes repair. Cadmium is a known animal teratogen and reproductive toxin. It has been shown to cause renal dysfunction in both humans and animals. Other toxic effects attributed to cadmium include immunosuppression (in animals), anemia (in humans), pulmonary disease (in humans), possible effects on the endocrine system, defects in sensory function, and bone damage. The oral LD_{50} in the rat was 225 mg/kg.

Toxicity to Wildlife and Domestic Animals

Laboratory experiments suggest that cadmium may have adverse effects on reproduction in fish at levels present in lightly to moderately polluted waters.

The acute LC_{50} for freshwater fish and invertebrates generally ranged from 100 to 1,000 µg/liter; salmonids are much more sensitive than other organisms. Saltwater species were in general 10-fold more tolerant to the acute effects of cadmium. Chronic tests have been performed and show that cadmium has cumulative toxicity and acute-chronic ratios that range of from 66 to 431. Bioconcentration factors were generally less than 1,000 but were as high as 10,000 for some freshwater fish species.

No adverse effects on domestic or wild animals were reported in the studies reviewed.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed 1984)

Freshwater

Acute toxicity: $e^{(1.30[\ln(\text{hardness})] - 3.92)}$ µg/liter

Chronic toxicity: $e^{(0.87[\ln(\text{hardness})] - 4.38)}$ µg/liter

Saltwater

Acute toxicity: 38 µg/liter
Chronic toxicity: 12 µg/liter

Human Health

Criterion: 10 µg/liter

CAG Unit Risk for inhalation exposure (USEPA): 7.8 $(\text{mg/kg/day})^{-1}$

Interim Primary Drinking Water Standard (USEPA): 10 µg/liter

NIOSH Recommended Standards: 40 $\mu g/m^3$ TWA
200 $\mu g/m^3$/15 min Ceiling Level.

OSHA Standards: 200 $\mu g/m^3$ TWA
600 $\mu g/m^3$ Ceiling Level

ACGIH Threshold Limit Values: 50 $\mu g/m^3$ TWA

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

CLEMENT ASSOCIATES, INC. 1983. Assessment of the Weight of Evidence for Risk Assessment for Four Selected Toxic Air Pollutants. Report Prepared for the Air Economic Branch, OPRM, U.S. Environmental Protection Agency. May 1983.

FLEISCHER, M., SAROFIM, A.F., FASSETT, D.W., HAMMOND, P., SCHARKETTE, H.T., NISBET, I.C.T., and EPSTEIN, S. 1974. Environmental impact of cadmium: A review by the panel on hazardous trace substances. Environ. Health Perspect. 7:253-323

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

TAKENAKA, S., OLDIGES, H., KONIG, H., HOCHRAINER, D., and OBERDORSTER, G. 1983. Carcinogenicity of cadmium chloride aerosols in W rats. JNCI 70:367-371

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Cadmium. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-025

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Cadmium. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H038 (Final Draft)

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1985. Health Assessment Document for Dichloromethane (Methylene Chloride). Office of Health and Environmental Assessment. Washington, D.C. February 1985. EPA 600/8-82/004F

MAGNESIUM

Summary

Exposure to magnesium oxide fumes can cause metal fume fever in humans. Exposure to magnesium oxide dust can irritate the eyes and respiratory tract. Ingestion of very high levels of magnesium salts can cause central nervous sytem effects; it can also have a laxative action.

Background Information

Magnesium is the eighth most abundant element on earth. It is very reactive chemically and does not occur uncombined in nature. Finely divided magnesium can react with water to yield hydrogen gas and magnesium hydroxide. However, reaction of solid magnesium with water is self-limiting because of the formation of a film of magnesium hydroxide. As a result, elemental magnesium is considered insoluble in water.

CAS Number: 7439-95-4

Chemical Formula: Mg

IUPAC Name: Magnesium

Chemical and Physical Properties

Atomic Weight: 24.312

Boiling Point: 1107°C

Melting Point: 648.8°C

Specific Gravity: 1.738

Solubility in Water: Insoluble; most salts are very soluble

Transport and Fate

Most magnesium salts are very soluble at pH levels normally found in natural waters, and the magnesium ion is readily transported in surface water, soil, and groundwater. The extent of magnesium transport in soil is dependent, in part, on the cation exchange capacity of the soil. Evaporation of ocean spray particles and subsequent atmospheric transport of magnesium

can occur. Atmospheric transport of dusts and fumes of compounds such as magnesim oxide can also occur.

Health Effects

There is no evidence to suggest that magnesium has carcinogenic, mutagenic, teratogenic, or reproductive effects in humans or experimental animals. Magnesium oxide fumes can produce metal fume fever in humans and experimental animals. Exposure to magnesium oxide dust may cause irritation of the eyes and respiratory tract. Human exposure to magnesium usually occurs by ingestion. Magnesium is an essential element for humans, animals, and plants. Ingestion of 3.6 to 4.2 mg/kg/day is thought to be adequate for maintenance of magnesium balance in humans. The average adult American is estimated to ingest 240 to 480 mg/kg/day in food and water. However, magnesium is absorbed relatively poorly by the gastrointestinal tract and also is readily excreted in the urine. Excessive magnesium retention in the body generally only occurs as a result of severe kidney disease. Symptoms of hypermagnesemia can include a sharp drop in blood pressure, and respiratory paralysis due to central nervous system depression. Ingestion of magnesium salts at concentrations over 700 mg/liter can have a laxative effect. However, humans can adapt to ingestion of these levels in a relatively short time. Magnesium has a very unpleasant taste in water at concentrations producing toxic effects.

Different magnesium compounds vary in the severity of their toxic effects to experimental animals. Such effects include central nervous system and purgative effects similar to those seen in humans. Subcutaneous injection of powdered magnesium or magnesium alloys can produce symptoms in experimental animals resembling gas gangrene. Application of powdered magnesium to abraded skin can produce an inflammatory reaction. However, these types of skin effects have not been reported in exposed workers.

Toxicity to Wildlife and Domestic Animals

Available data are not adequate to characterize the toxicity of magnesium to wildlife or domestic animals. Observed effects are generally related to deficiency symptoms.

Regulations and Standards

OSHA Standard: 15 mg/m^3 (magnesium oxide fume)

ACGIH Threshold Limit Values:

10 mg/m^3 TWA (magnesium oxide fume)
10 mg/m^3 TWA (magnesite, nuisance particulate)
20 mg/m^3 STEL (magnesite, nuisance particulate)

U.S. Department of Transportation: Flammable solid; dangerous when wet

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

CLAYTON, G.D., and CLAYTON, F.E., eds. 1981. Patty's Industrial Hygiene and Toxicology. Vol. 2A: Toxicology. 3rd rev. ed. John Wiley and Sons, New York. 2,878 pages

DOULL, J., KLAASSEN, C.D., and AMDUR, M.O., eds. 1980. Casarett and Doull's Toxicology: The Basic Science of Poisons. 2nd ed. Macmillan Publishing Co., New York. 778 pages

NATIONAL ACADEMY OF SCIENCES (NAS). 1977. Drinking Water and Health. Safe Drinking Water Committee, Washington, D.C. 939 pages

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1984. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. April 1984

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2332 pages

MANGANESE

Summary

Manganese chloride produced lymphomas and manganese sulfate, tumors after injestion into mice. In humans, chronic exposure to manganese causes degenerative changes in the central nervous system in the form of a Parkinson-like disease; liver changes also occur. Acute exposure causes manganese pneumonitis.

CAS Number: 7439-96-5

Chemical Formula: Mn

IUPAC Name: Manganese

Chemical and Physical Properties

Atomic Weight: 54.938

Boiling Point: 1962°C

Melting Point: 1244°C

Specific Gravity: 7.20

Solubility in Water: Decomposes; some compounds are soluble

Transport and Fate

Manganese occurs most commonly in the +2 and +4 oxidation states in aquatic systems. Its solubility depends to a great extent on pH, dissolved oxygen, and presence of complexing agents. In saltwater, it is estimated that 85% or more of the manganese present exists in a soluble form. In freshwater, manganese can occur as the soluble ion, in complex organic ions, or in colloidal suspensions. Manganese often occurs at higher concentrations near the bottom of stratified lakes because it can be released from sediments, as the manganous ion, under reducing conditions.

In the soil, the concentration and chemical form in which manganese occur can be affected by pH, cation exchange capacity, drainage, organic matter content, and other factors. The solubility of manganese is increased at low pH and under reducing conditions. The presence of high concentrations of chlorides, nitrates, or sulfates may also increase solubility. Under these conditions, manganese is more easily taken up by plants

or transported in aqueous solution. Lack of sufficient cation exchange sites, which are provided by organic matter or clay, can also result in greater leaching of manganese to surface or groundwater.

Atmospheric transport of manganese fumes or dusts can occur. These materials can be returned to the earth by wet or dry deposition.

Health Effects

There are no epidemiological studies suggesting that manganese or its compounds are carcinogenic or have teratogenic or reproductive effects in humans. Exposure to manganese chloride by intraperitoneal or subcutaneous routes was reported to cause lymphomas in mice. Manganese sulfate was found to produce tumors after intraperitoneal administration in mice. No other reports of unequivocal carcinogenic activity are available for common manganese compounds. Some manganese compounds, notably manganese chloride, have exhibited mutagenic activity in a variety of test systems. Manganese compounds do not appear to be teratogenic, however.

In humans, manganese dusts and compounds have relatively low oral and dermal toxicity, but they can cause a variety of toxic effects after inhalation exposure. Acute exposure to very high concentrations can cause manganese pneumonitis, increased susceptibility to respiratory disease, and pathologic changes including epithelial necrosis and mononuclear proliferation. Chronic manganese poisoning is more common, but generally occurs only among persons occupationally exposed to manganese compounds. Degenerative changes in the central nervous system are the major toxic effects. Early symptoms include emotional changes, followed by a masklike face, retropulsion or propulsion, and a Parkinson's-like syndrome. Liver changes are also frequently seen. Individuals with an iron deficiency may be more susceptible to chronic poisoning.

Duplication of human exposure symptoms in experimental animals has only been partially successful. In rabbits exposed by inhalation to manganese dust, manganese pneumonitis did not develop, but fibrotic changes in the lungs were observed. Central nervous system effects characteristic of chronic exposure in humans have only been reproduced in monkeys.

Toxicity to Wildlife and Domestic Animals

Adequate data for characterization of the toxicity of manganese to wildlife or domestic animals are not available.

A 48-hour LC_{50} value of 16 mg/liter of manganese is reported for embryos of the oyster Crassostrea virginica. For the softshell clam Mya arenaria a 168-hour LC_{50} value of 300 mg/liter is reported.

Regulations and Standards

OSHA Standard: 5 mg/m^3 Ceiling Level

ACGIH Threshold Limit Values:

1 mg/m^3 TWA (fume)
3 mg/m^3 STEL (fume)
5 mg/m^3 Ceiling Level (dust and compounds)

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

DOULL, J., KLAASSEN, C.D., and AMDUR, M.O., eds. 1980. Casarett and Doull's Toxicology: The Basic Science of Poisons. 2nd ed. Macmillan Publishing Co., New York. 778 pages

EISLER, R. 1977. Acute toxicities of selected heavy metals to the softshell clam, Mya arenaria. Bull. Environ. Contam. Toxicol. 17:137-145

NATIONAL ACADEMY OF SCIENCE (NAS). 1973. Medical and Biological Effects of Environmental Pollutants: Manganese. Washington, D.C. 191 pages

NATIONAL ACADEMY OF SCIENCES (NAS). 1977. Drinking Water and Health. Safe Drinking Water Committee, Washington, D.C. 939 pages

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Manganese. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H057 (Final Draft)

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2332 pages

CHROMIUM

Summary

Chromium is a heavy metal that generally exists in either a trivalent or hexavalent oxidation state. Hexavalent chromium (Cr VI) is rather soluble and is quite mobile in groundwater and surface water. However, in the presence of reducing agents it is rapidly converted to trivalent chromium (Cr III), which is strongly adsorbed to soil components and consequently is much less mobile. A number of salts of hexavalent chromium are carcinogenic in rats. In addition, an increased incidence of lung cancer was seen in workers occupationally exposed to chromium VI. Hexavalent chromium also causes kidney damage in animals and humans: Trivalent chromium is less toxic than hexavalent chromium; its main effect is contact dermatitis in sensitive individuals. The EPA Ambient Water Quality Criteria for the protection of human health are 50 µg/liter for Cr VI and 170 µg/liter for Cr III.

CAS Number: 7440-47-3

Chemical Formula: Cr

IUPAC Name: Chromium

Chemical and Physical Properties (Metal)

Atomic Weight: 51.996

Boiling Point: 2672°C

Melting Point: 1857 ± 20°C

Specific Gravity: 7.20 at 28°C

Solubility in Water: Insoluble; some compounds are soluble

Transport and Fate

Hexavalent Cr is quite soluble, existing in solution as a component of a complex anion. It is not sorbed to any significant degree by clays or hydrous metal oxides. The anionic form varies according to pH and may be a chromate, hydrochromate, or dichromate. Because all anionic forms are so soluble, they are quite mobile in the aquatic environment. Cr VI is efficiently

removed by activated carbon and thus may have some affinity for organic materials in natural water. Cr VI is a moderately strong oxidizing agent and reacts with reducing materials to form trivalent chromium. Most Cr III in the aquatic environment is hydrolyzed and precipitates as chromium hydroxide. Sorption to sediments and bioaccumulation will remove much of the remaining Cr III from solution. Cr III is adsorbed only weakly to inorganic materials. Cr III and Cr VI are readily interconvertible in nature depending on microenvironmental conditions such as pH, hardness, and the types of other compounds present. Soluble forms of chromium accumulate if ambient conditions favor Cr VI. Conditions favorable for conversion to Cr III lead to precipitation and adsorption of chromium in sediments.

In air, chromium is associated almost entirely with particulate matter. Sources of chromium in air include windblown soil and particulate emissions from industrial processes. Little information is available concerning the relative amounts of Cr III and Cr VI in various aerosols. Relatively small particles can form stable aerosols and can be transported many miles before settling out.

Cr III tends to be adsorbed strongly onto clay particles and organic particulate matter, but can be mobilized if it is complexed with organic molecules. Cr III present in minerals is mobilized to different extents depending on the weatherability and solubility of the mineral in which it is contained. Hexavalent compounds are not strongly adsorbed by soil components and Cr VI is mobile in groundwater. Cr VI is quickly reduced to CR III in poorly drained soils having a high content of organic matter. Cr VI of natural origin is rarely found in soils.

Health Effects

The hexavalent form of chromium is of major toxicological importance in higher organisms. A variety of chromate (Cr VI) salts are carcinogenic in rats and an excess of lung cancer has been observed among workers in the chromate-producing industry. Cr VI compounds can cause DNA and chromsome damage in animals and humans, and Cr (VI) trioxide is teratogenic in the hamster. Inhalation of hexavalent chromium salts causes irritation and inflammation of the nasal mucosa, and ulceration and perforation of the nasal septum. Cr VI also produces kidney damage in animals and humans. The liver is also sensitive to the toxic effects of hexavalent Cr, but apparently less so than the kidneys or respiratory system. Cr III is less toxic than Cr VI; its main effect in humans is a form of contact dermatitis in sensitive individuals.

Cr III:

Aquatic Life (Proposed Criteria)

Freshwater

Acute toxicity: $e^{(0.819[\ln(\text{hardness})]+3.568)}$ µg/liter

Chronic toxicity: $e^{(0.819[\ln(\text{hardness})])+0.537)}$ µg/liter

Saltwater

The available data are not adequate for establishing criteria.

Human Health

Criterion: 170 µg/liter

CAG Unit Risk for inhalation exposure to CR VI (USEPA): 41 $(mg/kg/day)^{-1}$

National Interim Primary Drinking Water Standard: 50 µg/liter

NIOSH Recommended Standards for CR VI: 1 $\mu g/m^3$ carcinogenic
25 $\mu g/m^3$ noncarcinogenic TWA
50 $\mu g/m^3$ noncarcinogenic (15-min sample)

OSHA Standards: OSHA air standards have been set for several chromium compounds. Most recognized or suspected carcinogenic chromium compounds have ceiling limits of 100 $\mu g/m^3$.

ACGIH Threshold Limit Values: Several chromium compounds have TWAs ranging from 0.05 to 0.5 mg/m^3. Chromite ore processing (chromate), certain water insoluble Cr VI compounds, and chromates of lead and zinc are recognized or suspected human carcinogens and have 0.05 mg/m^3 TWAs.

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

INTERNATIONAL AGENCY FOR RESEARCH ON CANCER (IARC). 1980. IARC Monograph on the Evaluation of the Carcinogenic Risk of Chemicals to Humans. Vol. 23: Some Metals and Metallic Compounds. World Health Organization, Lyon, France

Cyanides move rather freely in soils but biodegradation would probably significantly decrease the amount present in the groundwater. Volatilization of HCN and nitriles may occur from soil surfaces.

Health Effects

Hydrogen cyanide and its simple salts, such as sodium cyanide, are highly toxic by all routes. Many reports are available regarding acute poisoning in humans. Hydrogen cyanide vapor is irritating at very low concentrations, is considered dangerous at 20 ppm (20 mg/m^3), and is fatal at concentrations of 100 ppm (100 mg/m^3) for one hour. NIOSH notes reports of chronic poisoning resulting in fatigue, weariness and other subjective symptoms in workers, but these findings have been disputed by other investigators. Chronic exposure to low levels of cyanide salts has been reported to cause enlargement of the thyroid gland in humans, apparently due to inefficient elimination of the cyanide metabolite thiocyanate. NIOSH (1976) concluded that there was no evidence of carcinogenicity, mutagenicity, or teratogenicity for cyanides. Cyanide has been shown to produce chromosome breaks in a plant, Vicia faba. Because of its mechanism of action, inhibition of the election transport system in oxidative phosphorylation, cyanide is acutely toxic to almost all forms of life. A reduction in the TLV for HCN from 10 mg/m^3 to a ceiling value of 3 mg/m^3 has been recommended by several investigators, to prevent the various nonspecific effects noted by several investigators (ACGIH 1980).

Toxicity to Wildlife and Domestic Animals

Cyanide is acutely toxic to both freshwater and saltwater organisms, causing death at levels of about 50 µg/liter in sensitive species and being fatal to many species at levels above 200 µg/liter. Final acute values were determined to be 44.7 µg/liter for freshwater species and 2.03 µg/liter for saltwater species. Effects such as reduced survival and reduced reproduction were seen in fish chronically exposed to free cyanide concentrations of from 10 to 50 µg/liter. The final acute chronic ratios were determined to be 10.7 and 3.5 for freshwater and saltwater organisms, respectively. The final chronic values were determined by dividing the acute values by the acute-chronic ratio, and were determined to be 4.2 for freshwater species and 0.57 for saltwater organisms. An accidental spill of cyanide caused the death of 4,800 fish in Oak Ridge, Tennessee. The long-term effects of this spill were not reported. Livestock death and environmental damage were caused by high levels of cyanide leaching from a drum disposal site in Illinois.

Summary

Cyanide can be present in many forms in the environment. The transport, fate, and toxicity of the chemical is quite dependent on the specific form. Hydrogen cyanide and its simple salts are highly toxic following acute exposure by humans, experimental animals, and both aquatic and terrestrial wildlife.

Background Information

Cyanide (CN-) is usually defined as hydrogen cyanide (HCN) and its salts. The chemical/physical properties, transport and fate, and toxicity of cyanide are quite dependent on the form of cyanide present.

CAS Number: 151-50-8; 143-33-9

Chemical Formula: CN-

IUPAC Name: Cyanide

Chemical and Physical Properties

Molecular Weight: 27 (HCN)

Boiling Point: 26.7°C (HCN)

Melting Point: -14°C (HCN)

Specific Gravity: 0.699 at 22°C (HCN)

Solubility in Water: Soluble (HCN)

Solubility in Organics: Soluble in alcohol and ether

Vapor Pressure: 657.8 mm Hg at 21.9°C (HCN)

Transport and Fate

The transport and fate of cyanide in the environment is dependent on the chemical compound containing the cyanide. Most free cyanide will be HCN in aquatic environments and will probably evaporate, although biodegradation is another possible fate process. Metal cyanides are generally insoluble and for

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed)

Freshwater

Acute toxicity: 22 µg/liter
Chronic toxicity: 4.2 µg/liter

Saltwater

Acute toxicity: 1.0 µg/liter
Chronic toxicity: 0.57 µg/liter

Human Health

Criterion: 200 µg/liter

Primary Drinking Water Standard (USEPA): 200 µg/liter

ACGIH Threshold Limit Value: 5 mg/m^3 TWA

REFERENCES

AMERICAN COUNCIL OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1976. Criteria for a Recommended Standard--Occupational Exposure to Hydrogen Cyanide and Cyanide Salts (NaCN, KCN, and $Ca(CN)_2$). Washington, D.C. DHEW Publication No. (NIOSH) 77-108

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1983

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Cyanides. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-037

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). [illegible]
Section B of Ambient Water Criteria for Cyanide--Aquatic Toxicology. Draft Report. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. August 1983

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Cyanide. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H011 (Final Draft)

VERSCHUEREN, K. 1977. Handbook of Environmental Data on Organic Chemicals. Van Nostrand Reinhold Co., New York. 659 pages

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2,332 pages

Summary

In the absence of photolytic degradation, dimethylnitrosamine is probably persistent in the environment. Dimethylnitrosamine is carcinogenic and produces lung, liver, and kidney tumors in rats and mice and liver tumors in several other animal species. It also exhibits transplacental carcinogenicity in animals and is mutagenic and embryotoxic. Both acute and chronic exposure have adverse effects on the liver in humans and experimental animals.

CAS Number: 62-75-9

Chemical Formula: $(CH_3)_2NNO$

IUPAC Name: n-Nitrosodimethylamine

Important Synonyms and Trade Names: n-Methyl-n-nitrosomethanamine, n,n-dimethylnitrosamine, DMN, DMNA, NDMA

Chemical and Physical Properties

Molecular Weight: 74.1

Boiling Point: 151°C

Specific Gravity: 1.0 at 20°C

Solubility in Water: Soluble in all proportions

Solubility in Organics: Soluble in organic solvents, lipids

Log Octanol/Water Partition Coefficient: 0.06 to -0.69

Transport and Fate

The most probable environmental fate of dimethylnitrosamine in aqueous solution appears to be slow photolytic degradation. Furthermore, although supporting data are limited, it has been speculated that hydrogen bonding of dimethylnitrosamine with humic acids or coordination with metal cations produces a photolabile intermediate and could lead to moderately rapid degradation in surface waters. Dimethylnitrosamine has been detected

in the atmosphere of metropolitan areas and near manufacturing facilities emitting this compound, suggesting that some atmospheric transport can occur. However, it is reported that photolytic degradation in air would be rapid, with a half-life of less than 1 hour. Airborne concentrations in excess of a few parts per billion appear to be unlikely except near sources of direct emissions. There is no evidence to suggest that oxidation or hydrolysis are important environmental fates.

Dimethylnitrosamine is completely miscible in water and is reported to be highly solvated. This information, along with limited experimental data, suggest that volatilization from surface waters is probably not an important process. Dimethylnitrosamine has a log octanol/water partition coefficient near 0; significant sorption by organic particulates is therefore unlikely. Experimental evidence confirms this and further suggests that sorption by clay particulates in wet soil is also unlikely. Because dimethylnitrosamine is completely miscible in water and has a low log octanol/water partition coefficient, bioaccumulation is probably an insignificant process. Although biodegradation in surface waters does not appear to be an important environmental fate, slow degradation of dimethylnitrosamine in sewage and soil is reported to occur. Based on this information, it is likely that in the absence of photolytic degradation dimethylnitrosamine would be very persistent in the environment.

Health Effects

Dimethylnitrosamine is considered to be carcinogenic in many experimental animal species by various routes of exposure. Dose-response relationships have been established in several studies. This compound produces liver, lung, and kidney tumors in some species of mice and rats after oral and inhalation exposure. Increased incidences of liver tumors have also been observed in many other animal species after oral administration. Inhalation exposure in rats has produced tumors of the ethroturbinals and nasal cavity. Although insufficient epidemiologic evidence exists to establish a causative role for dimethylnitrosamine in human carcinogenesis, IARC and other public health organizations recommend that this compound be regarded as a human carcinogen.

Dimethylnitrosamine is mutagenic in many microbial test systems with metabolic activation and in several other in vivo and in vitro test systems. This compound is reported to exhibit transplacental carcinogenicity and to be embryotoxic. No teratogenic effects have been reported. Acute and chronic exposure of humans and experimental animals to dimethylnitrosamine resulted primarily in a variety of hepatotoxic effects. In rats, an oral LD_{50} value of 40 mg/kg and an inhalation LD_{50} value of 37 mg/kg are reported.

Toxicity to Wildlife and Domestic Animals

In crayfish exposed to dimethylnitrosamine in water for 6 months, extensive antennal gland degeneration was observed at 200,000 µg/liter and hyperplasia of hepatopancreas tubular cells at 100,000 µg/liter. Rainbow trout fed dimethylnitrosamine for 52 weeks showed a dose-related increase in hepatocellular carcinoma at doses of 200, 400, and 800 mg/kg. The weighted average bioconcentration factor for the edible portion of all freshwater and estuarine aquatic organisms consumed by Americans is 0.026.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life

Freshwater

Acute toxicity: The available data for nitrosamines in general indicate that toxic effects occur at concentrations as low as 5,850 µg/liter and would occur at lower concentrations among species that are more sensitive than those tested.

Chronic toxicity: No available data

Saltwater

Acute toxicity: The available data for nitrosamines in general indicate that toxic effects occur at concentrations as low as 3,300,000 µg/liter and would occur at lower concentrations among species that are more sensitive than those tested.

Chronic toxicity: No available data

Human Health

Estimates of the carcinogenic risks associated with lifetime exposure to various levels of dimethylnitrosamine in water are:

Risk	Concentration
10^{-5}	14 ng/liter
10^{-6}	1.4 ng/liter
10^{-7}	0.14 ng/liter

ACGIH Threshold Limit Value: Suspected human carcinogen

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

INTERNATIONAL AGENCY FOR RESEARCH ON CANCER (IARC). IARC Monographs on the Evaluation of Carcinogenic Risk of Chemicals to Humans. Vol. 17: Some N-Nitroso Compounds. World Health Organization, Lyon, France. Pp. 125-175

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1984. Registry of Toxic Effects of Chemical Substances. Data Base. Washington, D.C. October 1984

SAX, N.I. 1975. Dangerous Properties of Industrial Materials. 4th ed. Van Nostrand Reinhold Co., New York. 1,258 pages

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Nitrosamines. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-064

Solubility in Organics: Depends on chemical species

Vapor Pressure: 0.0012 mm Hg at 20°C

Transport and Fate

Mercury and certain of its compounds, including several inorganic species and dimethyl mercury, can volatilize to the atmosphere from aquatic and terrestrial sources. Volatilization is reduced by conversion of metallic mercury to complexed species and by deposition of HgS in reducing sediments, but even so atmospheric transport is the major environmental distribution pathway for mercury. Precipitation is the primary mechanism for removal of mercury from the atmosphere. Photolysis is important in the breakdown of airborne mercurials and may be important in some aquatic systems. Adsorption onto suspended and bed sediments is probably the most important process determining the fate of mercury in the aquatic environment. Sorption is strongest into organic materials. Mercury in soils is generally complexed to organic compounds.

Virtually any mercury compound can be remobilized in aquatic systems by microbial conversion to methyl and dimethyl forms. Conditions reported to enhance biomethylation include large amounts of available mercury, large numbers of bacteria, the absence of strong complexing agents, near neutral pH, high temperatures, and moderately aerobic environments. Mercury is strongly bioaccumulated by numerous mechanisms. Methylmercury is the most readily accumulated and retained form of mercury in aquatic biota, and once it enters a biological system it is very difficult to eliminate.

Health Effects

When administered by intraperitoneal injection, metallic mercury produces implantation site sarcomas in rats. No other studies were found connecting mercury exposure with carcinogenic effects in animals or humans. Several mercury compounds exhibit a variety of genotoxic effects in eukaryotes. In general, organic mercury compounds are more toxic than inorganic compounds. Although brain damage due to prenatal exposure to methylmercury has occurred in human populations, no conclusive evidence is available to suggest that mercury causes anatomical defects in humans. Embryotoxicity and teratogenicity of methylmercury has been reported for a variety of experimental animals. Mercuric chloride is reported to be teratogenic in experimental animals. No conclusive results concerning the teratogenic effects of mercury vapor are available.

MERCURY

Summary

Both organic and inorganic forms of mercury are reported to be teratogenic and embryotoxic in experimental animals. In humans, prenatal exposure to methylmercury has been associated with brain damage. Other major target organs for organic mercury compounds in humans are the central and peripheral nervous system and the kidney. In animals, toxic effects also occur in the liver, heart, gonads, pancreas, and gastrointestinal tract. Inorganic mercury is generally less acutely toxic than organic mercury compounds, but it does affect the central nervous system adversely. The EPA Ambient Water Quality Criterion for the protection of human health is 144 ng/liter.

Background Information

Several forms of mercury, including insoluble elemental mercury, inorganic species, and organic species, can exist in the environment. In general, the mercurous (+1) salts are much less soluble than the more commonly found mercuric (+2) salts. Mercury also forms many stable organic complexes that are generally much more soluble in organic liquids than in water. The nature and solubility of the chemical species that occur in an environmental system depend on the redox potential and the pH of the environment.

CAS Number: 7439-97-6

Chemical Formula: Hg

IUPAC Name: Mercury

Chemical and Physical Properties (Metal)

Atomic Weight: 200.59

Boiling Point: 356.58°C

Melting Point: -38.87°C

Specific Gravity: 13.5939 at 20°C

Solubility in Water: 81.3 µg/liter at 30°C; some salts and organic compounds are soluble

Chronic dietary exposure of chickens to mercuric chloride at growth inhibitory levels causes immune suppression, with a differential reduction effect on specific immunoglobulins.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life (Proposed Criteria)

Freshwater

Acute toxicity: 1.1 µg/liter
Chronic toxicity: 0.20 µg/liter

Saltwater

Acute toxicity: 1.9 µg/liter
Chronic toxicity: 0.10 µg/liter

Human Health

Criterion: 144 ng/liter

Primary Drinking Water Standard: 0.002 mg/liter

NIOSH Recommended Standard: 0.05 mg/m^3 TWA (inorganic mercury)

OSHA Standard: 0.1 mg/m^3 Ceiling Level

ACGIH Threshold Limit Values:

0.01 mg/m^3 TWA (alkyl compounds)
0.03 mg/m^3 STEL (alkyl compounds)
0.05 mg/m^3 TWA (vapor)
0.1 mg/m^3 TWA (aryl and inorganic compounds)

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

BRIDGER, M.A., and THAXTON, J.P. 1983. Humoral immunity in the chicken as affected by mercury. Arch. Environ. Contam. Toxicol. 12:45-49

In humans, alkyl mercury compounds pass through the blood brain barrier and the placenta very rapidly, in contrast to inorganic mercury compounds. Major target organs are the central and peripheral nervous systems, and the kidney. Methylmercury is particularly hazardous because of the difficulty of eliminating it from the body. In experimental animals, organic mercury compounds can produce toxic effects in the gastrointestinal tract, pancreas, liver, heart, and gonads, with involvement of the endocrine, immunocompetent, and central nervous systems.

Elemental mercury is not highly toxic as an acute poison. However, inhalation of high concentrations of mercury vapor can cause pneumonitis, bronchitis, chest pains, dyspnea, coughing, stomatitis, gingivitis, salivation, and diarrhea. Soluble mercuric salts are highly poisonous on ingestion, with oral LD_{50} values of 20 to 60 mg/kg reported. Mercurous compounds are less toxic when administered orally. Acute exposure to mercury compounds at high concentrations causes a variety of gastrointestinal symptoms and severe anuria with uremia. Signs and symptoms associated with chronic exposure involve the central nervous system and include behavioral and neurological disturbances.

Toxicity to Wildlife and Domestic Animals

The toxicity of mercury compounds has been tested in a wide variety of aquatic organisms. Although methylmercury appears to be more toxic than inorganic mercuric salts, few acute or chronic toxicity tests have been conducted with it. Among freshwater species, the 96-hour LC_{50} values for inorganic mercuric salts range from 0.02 µg/liter for crayfish to 2,000 µg/liter for caddisfly larvae. Acute values for methylmercuric compounds and other mercury compounds are only available for fishes. In rainbow trout, methylmercuric chloride is about ten times more toxic to rainbow trout than mercuric chloride, which is acutely toxic at about 300 µg/liter at 10°C. Methylmercury is the most chronically toxic of the tested compounds, with chronic values for Daphnia magna and brook trout of 1.00 and 0.52 µg/liter, respectively. The acute-chronic ratio for Daphnia magna is 3.2.

Mean acute values for saltwater species range from 3.5 to 1,680 µg/liter. In general, molluscs and crustaceans are more sensitive than fish to the acute toxic effects of mercury. A life-cycle experiment with the mysid shrimp showed that inorganic mercury at a concentration of 1.6 µg/liter significantly influences time of appearance of first brood, time of first spawn, and productivity. The acute-chronic ratio for the mysid shrimp is 2.9.

NATIONAL INSTITUTE FOR OCCUPATIONAL SAFETY AND HEALTH (NIOSH). 1983. Registry of Toxic Effects of Chemical Substances Data Base. Washington, D.C. October 1983

SHEPARD, T.H. 1980. Catalog of Teratogenic Agents. 3rd ed. Johns Hopkins University Press, Baltimore. 410 pages

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Mercury. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-058

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Water quality criteria; Request for comments. Fed. Reg. 49: 4551-4553

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Mercury. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H042 (Final Draft)

WEAST, R.E., ed. 1981. Handbook of Chemistry and Physics. 62nd ed. CRC Press, Cleveland, Ohio. 2,332 pages

WORLD HEALTH ORGANIZATION (WHO). 1976. Environmental Health Criteria: 1. Mercury. World Health Organization, Geneva. 131 pages

POLYCHLORINATED BIPHENYLS

Summary

Polychlorinated biphenyls (PCBs) are very persistent in the natural environment and are readily bioaccumulated. In humans, exposure to PCBs has been associated with chloracne, impairment of liver function, a variety of neurobehavioral symptoms, menstrual disorders, minor birth abnormalities, and an increased incidence of cancer. Experimental animals exposed to PCBs experienced an increased incidence of cancer; reproductive problems; neurobehavioral degradation; pathological changes in the liver, stomach, skin, and other organs; and suppression of immunological function. PCBs are often contaminated, and these contaminants may be much more toxic than the PCBs themselves. The EPA Ambient Water Quality Criterion for the protection of human health corresponding to an excess cancer risk of 10^{-6} is 0.079 ng/liter.

Background Information

Polychlorinated biphenyls (PCBs) are complex mixtures of chemicals composed of two connected benzene rings with 1 to 10 chlorine atoms attached. The chemical, physical, and biological properties of these materials depend to a large degree on the amount and location of the chlorine atoms on the two benzene rings of each specific PCB and on the particular mixture of individual chlorobiphenyls that comprise the mixture.

CAS Number: 1336-36-3

Chemical Formula: $C_6H_5Cl_xC_6H_5Cl_x$

IUPAC Name: Specific for each polychlorinated biphenyl

Important Synonyms and Trade Names: PCBs, chlorinated biphenyls, polychlorobiphenyls, Aroclor, Kanechlor, Clophen

Chemical and Physical Properties

Molecular Weight: 189-399*

Boiling Point: 267°C and up*

Melting Point: 54-310°C*

---

*Increases with increasing chlorination.

Specific Gravity: 1.3 to 1.5 at 20°C*

Solubility in Water: 0.003-0.6 mg/liter

Solubility in Organics: Soluble in most common organic solvents

Log Octanol/Water Partition Coefficient: 4-6*

Vapor Pressure: 10^{-3}-10^{-5} mm Hg at 20°C**

Henry's Law Constant: 10^{-3} to -10^{-5} atm m^3/mole

Transport and Fate

The transport and fate of polychlorinated biphenyls has been studied extensively, and although individual chemicals vary in the rates at which processes occur, some generalizations can be made about PCBs as a class. PCBs are relatively inert, and therefore persistent, compounds, with low vapor pressures, low water solubility, and high log octanol/water partition coefficients. Despite their low vapor pressures, they have a high activity coefficient in water, which causes a higher rate of volatilization than might normally be expected. Volatilization and persistence account for the ubiquitous nature of PCBs in the environment. Adsorption to the organic material in soil or sediments is probably the major fate of at least the more heavily chlorinated PCBs. Once bound, the PCBs may persist for years with slow desorption providing continuous, low-level exposure to the surrounding locality. Bioaccumulation of PCBs also occurs, with most of the compound stored in the adipose tissue of the body. PCBs are degraded primarily by two routes. Less heavily chlorinated PCBs (mainly the mono-, di-, and trichlorinated PCBs) can be biodegraded by some soil microorganisms. PCBs with five or more chlorines are not measurably biodegraded. These heavier PCBs can be photolyzed by ultraviolet light. This process is extremely slow, but it may be the most important degradation process for these very persistent compounds.

Assessing the toxicity of PCBs is complicated by the fact that several different mixtures have been produced and distributed commercially and by the presence of highly toxic contaminants in some commercial mixtures. Some of these contaminants can be formed by combustion of PCBs or even by high-temperature treatment in service, so that used materials may be more toxic than the commercial mixtures whose toxicity has been studied.

---

*Increases with increasing chlorination.
**Decreases with increasing chlorination.

Health Effects

In humans exposed to PCBs (in the workplace or via accidental contamination of food), reported adverse effects include chloracne (a long-lasting, disfiguring skin disease), impairment of liver function, a variety of neurobehavioral and affective symptoms, menstrual disorders, minor birth abnormalities, and probably increased incidence of cancer. Animals experimentally exposed to PCBs have shown most of the same symptoms, as well as impaired reproduction; pathological changes in the liver, stomach, skin, and other organs; and suppression of immunological functions. PCBs are carcinogenic in rats and mice and, in appropriate circumstances, enhance the effects of other carcinogens. Reproductive and neurobiological effects of PCBs have been reported in rhesus monkeys at the lowest dose level tested, 11 µg/kg body weight/day over a period of several months.

Toxicity to Wildlife and Domestic Animals

Polychlorinated biphenyls are bioaccumulated and can be biomagnified. Therefore, their toxicity increases with length of exposure and position of the exposed species on the food chain. The toxicity of the various PCB mixtures is also dependent on their composition. Because of the complexity of PCB toxicity, only general effects will be discussed here.

The 96-hour LC_{50} values for rainbow trout, bluegills, and channel catfish were around 20 mg/liter. The same species exposed for 10 to 20 days had LC_{50} values of about 0.1 mg/liter. Invertebrate species were also adversely affected, with some species having 7-day LC_{50} values as low as 1 µg/liter. In general, juvenile organisms appeared more susceptible to the effects of PCBs than either eggs or adults.

Three primary ways in which PCBs can affect terrestrial wildlife are outright mortality, adversely affecting reproduction, and changing behavior. PCB doses greater than 200 ppm in the diet or 10 mg/kg body weight (bw) caused some mortality in sensitive bird species exposed for several days. Doses around 1,500 ppm (diet) or about 100 mg/kg (bw) caused extensive mortality in these sensitive species. They generally caused some mortality in all species, with the level being dependent on the length of exposure and the particular PCB mixture. Some mammalian species are especially susceptible to PCBs. For example, mink died when fed as little as 5 ppm in the diet (equivalent to less than 1 mg/kg bw/day). PCBs caused lower egg production; deformities; decreased hatchability, growth, and survival; and some eggshell thinning in reproductive studies on chickens fed doses of 20 ppm in the diet (1 mg/kg bw). Mink fed 1 ppm in the diet (0.2 mg/kg bw) had lower reproductive success, and there are indications that an increased incidence

of premature births in some marine mammals was linked to PCB exposure. Behavioral effects on wildlife include increased activity, decreased avoidance response, and decreased nesting, all of which could significantly influence survival in the wild.

No toxic effects on domestic animals other than chickens were reported in the sources reviewed, but susceptible species would probably be affected in a similar manner to laboratory animals and wildlife.

Regulations and Standards

Ambient Water Quality Criteria (USEPA):

Aquatic Life

Freshwater

Acute toxicity: 2 µg/liter
Chronic toxicity: 0.014 µg/liter

Saltwater

Acute toxicity: 10 µg/liter
Chronic toxicity: 0.030 µg/liter

Human Health

Estimates of the carcinogenic risks associated with lifetime exposure to various concentrations of PCBs in water are:

Risk	Concentration
10^{-5}	0.79 ng/liter
10^{-6}	0.079 ng/liter
10^{-7}	0.0079 ng/liter

CAG Unit Risk (USEPA): 4.34 $(mg/kg/day)^{-1}$

NIOSH Recommended Standard: 1.0 $\mu g/m^3$ TWA

ACGIH Threshold Limit Value: 0.5 mg/m^3 TWA

REFERENCES

AMERICAN CONFERENCE OF GOVERNMENTAL INDUSTRIAL HYGIENISTS (ACGIH). 1980. Documentation of the Threshold Limit Values. 4th ed. Cincinnati, Ohio. 488 pages

INTERNATIONAL AGENCY FOR RESEARCH ON CANCER (IARC). 1978. IARC Monographs on the Evaluation of Carcinogenic Risk of Chemicals to Humans. Vol. 18: Polychlorinated Biphenyls and Polybrominated Biphenyls. World Health Organization, Lyon, France. Pp. 43-103

NATIONAL ACADEMY OF SCIENCES (NAS). 1977. Drinking Water and Health. Safe Drinking Water Committee, Washington, D.C. 939 pages

ROBERTS, J.R., RODGERS, D.W., BAILEY, J.R., and RORKE, M.A. 1978. Polychlorinated Biphenyls: Biological Criteria for an Assessment of their Effects on Environmental Quality. National Research Council of Canada, Ottawa, Canada. NRCC No. 16077

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1976. National Conference on Polychlorinated Biphenyls (November 19-21, 1975, Chicago, Illinois). Office of Toxic Substances, Washington, D.C. March 1976. EPA 560/6-75-004

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1979. Water-Related Environmental Fate of 129 Priority Pollutants. Washington, D.C. December 1979. EPA 440/4-79-029

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1980. Ambient Water Quality Criteria for Polychlorinated Biphenyls (PCBs). Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. October 1980. EPA 440/5-80-054

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1984. Health Effects Assessment for Polychlorinated Biphenyls. Environmental Criteria and Assessment Office, Cincinnati, Ohio. September 1984. ECAO-CIN-H004 (Final Draft)

U.S. ENVIRONMENTAL PROTECTION AGENCY (USEPA). 1985. Health Assessment Document for Dichloromethane (Methylene Chloride). Office of Health and Environmental Assessment. Washington, D.C. February 1985. EPA 600/8-82/004F

FEB 18 1987

Dick Ruelle
U.S. Fish & Wildlife Service
1830 Second Avenue
Rock Island, Illinois 61201

Re: Crab Orchard Lake RI/FS

Dear Mr. Ruelle:

Attached is a tabulation of QA/QC comments on the Phase I data and recommended date useability. The QA/QC comments are taken from memos and discussions with personnel from the U.S. Environmental Protection Agency, Region V, Contract Laboratory Management Section (CPMS) and the Quality Assurance Office (QAO).

The recommended data useability is a concurrence of opinion between myself and personnel from the CPMS and QAO. If you or your contractors disagree with these recommendations, please contact me.

My understanding is that if data is considered not useable, it should not even be included in Remedial Investigation Report. If data is useable for screening only, this should be identified in the Report and the data should be separated from data that is considered quantitative. Other data qualifiers should also be identified in the Report.

It should be noted that specific additional documentation needed to validate the data are identified for the ETC CLP organics and Weston explosives. If this missing documentation is submitted, it is possible that this data will be determined to be more useable.

Sincerely,

ORIGINAL SIGNED BY
RICHARD BOICE

Richard E. Boice
Remedial Site Project Manager

cc: B. Cowles, IEPA
D. Iyer, O'Brien & Gere

Laboratory	Parameters	QA/QC Problems[1] (Reference)	Recommended Data Usability
O'Brien & Gere	CLP Organics Volatiles	No tuning data Mass ratio outside limits for brominated cmpds. Matrix spikes outside for some compounds. (2/22/86 memo from Pat Churilla)	Not useable for: 2-butanone, vinyl acetate, 4-methyl-2-pentanone. Positive values are estimates. Negatives are not useable unless nothing detected in whole fraction.
	BNA's	Tuning data unacceptable. Compounds not found in initial calibration. Calibration data missing on two dates. Low recovery on some samples. (8/11/86 memo from Patrick Churilla)	Not useable for: aniline, bis (2-chloro-isopropyl) ether, 4-chloroaniline, 2-nitro-sodiphenylamine, benzidene, 3,3-dichloro-benzidine, di-n-octyl-phthalate, benzo(a)pyrene, indeno (1,2,3) pyrene, dibenzo(ah) anthracene. Positive values are estimates, Negatives are not useable unless nothing detected in whole fraction.
	Pesticide/PCBs	Retension time shift. (4/24/86 and 10/16/86 memos from Patrick J. Churilla)	Useable for screening purposes. PCB results are qualitative. (p. 2, 9/26/86 from James Adams)
	GC screening of soils	Data not assessed in detail	Useable for screening. PCB results are qualitative. (p. 2 of 9/26/86 memo from James Adams)

Footnote:

(1) Chain-of-custody procedures were not followed in the laboratory for any Phase I samples (p. 2, 7/3/86 memo from James Adams).

Laboratory	Parameters	QA/QC Problems (Reference)	Recommended Data Useability
O'Brien & Gere	PCB's	(9/26/86 memo from James Adams)	Useable
	Atomic Absorption Screen (Ag, As, Be, Cd, Cu, Ni, Pb, Se, Zn)	No raw data and very little QA/QC documentation is available. Results appear to be inconsistent with results from previous samplings. (9/26/86 memo from James Adams)	Not useable
	Mercury	Very high blank and poor calibration (8/1/86 memo from Jay Thakkar; 9/[illegible]'86 memo from James Adams)	Not useable
	TKN, total phosphorus	Strip charts not matched with raw data. Duplicates and spikes out of control. (8/1/86 memo from Jay Thakkar)	Values estimated. Useable only for screening relative values.
	Cyanide	High detection limit. No laboratory QC performed. (8/1/86 memo from Jay Thakkar, 7/3/86 memo from James Adams)	Not useable
	Water extractable nitrate-nitrite	Some strip charts very noisy. Wrong reading from strip charts (8/1/86 memo from Jay Thakkar)	Values estimated. Useable only for screening relative values.
	Water extractable TOC, ammonia, chloride spec cond., pH, sulfate	No records reviewed.	Probably useable for screening of relative concentrations only.
	TOX in water	No records reviewed.	Probably useable for screening of relative concentrations only.
	TOX in soils	Method not determined.	Not determined.

Laboratory	Parameters	QA/QC Problems (Reference)	Recommended Data Useability
ETC	ICP metals	(8/1/86 memo from Jay Thakkar)	Estimated values. Useable for screening purposes only.
	CLP Organics	No calibration data for VOA's & BNA's. No raw data on spikes and method blanks. No duplicate spike. BNA contained TIC's that could cause false positives. (8/11/86 memo from Patrick Churilla)	PCB results useable; BNA & Volatiles are estimated and useable for screening only. Pesticide-not determined because no documentation provided.
	Dioxin/Furan	Diphenylethers not checked as interference (8/11/86 memo from Patrick Churilla)	Useable
Weston	Explosive residues	Detection limits should be 5x higher. No retention time windows. Percent moisture not reported. Method of calculation not given. Operating conditions not reported. (7/16/86 memo from Patrick Churilla)	Useable with detection limit 5 x detection limit reported.
Hazelton	CLP Organics	(1/14/87 memo from Patrick Churilla)	Volatiles, BNA & PCB's useable except that where acetone, methylene chloride or phthalates detected in blank, there may be false positives and other positive values will be estimated and biased high. Pesticide analyses should be considered qualitative.
	ICAP metals	Not assessed	

Laboratory	Parameters	QA/QC Problems (Reference)	Recommended Data Usability
	Cyanide	Not assessed	
	Mercury	Not assessed	
	TKN, ammonia, phosphorus	Not assessed	
Upstate Laboratories	Arsenic, selenium	Not assessed	

22:23

BIOLOGICAL DATA RELEVANT TO THE EVALUATION OF CARCINOGENIC RISK TO HUMANS

Prepared for
Scientific Advisory Panel,
Safe Drinking Water and Toxic Enforcement Act
State of California

Prepared by
Dr. Raymond D. Harbison
Dr. Robert C. James
Dr. Stephen M. Roberts

Division of Interdisciplinary Toxicology
University of Arkansas
School of Medicine
Little Rock, Arkansas
U.S.A.

August 1987

TABLE OF CONTENTS

EXECUTIVE SUMMARY

Evaluation of Animal Data

The investigation of PCBs' carcinogenic potential in mice is limited to two short studies, while some eight to ten studies have been reported using various strains of rat. The PCB mixtures tested thus far are : 1) in mice - Kanechlor 300, Kanechlor 400, Kanechlor 500 and Aroclor 1254; and 2) in rats - Kanechlor 300, Kanechlor 400, Kanechlor 500, Clophen A30, Clophen A60, Aroclor 1254 and Aroclor 1260.

PCBs have been found to be tumorigenic in mice with Aroclor 1254 producing hepatomas after 11 months of exposure and Kanechlor 500 (similar in composition to Aroclor 1254) inducing hepatocellular carcinomas after 8 months of exposure. These lesions were shown to be reversible and specific for the dose (500 ppm) and chlorination of the PCB mixture.

In rats, Aroclor 1260 or its equivalent, Clophen A60, have produced hepatocellular carcinomas in three studies at doses of approximately 100 ppm, a dose which appears to represent the maximally tolerated dose for rats. A review of these three studies indicates that the tumors occur very late in the life of the animal, with a significant incidence of tumors only beginning to appear after about two years of exposure. Of interest is the fact that in all three studies the PCB treatment, while increasing the incidence of liver cancer, did not increase the total tumor incidence. The total tumor incidence was not increased in these studies because in each case the incidence of other tumor types had been significantly decreased. This suggestion of antitumor activity of PCBs has also been demonstrated in a study examining the effect of PCB exposure on the final tumor incidence in animals following the transplantation of the Walker 256 sarcoma. The effects of chronic PCB-treatment was not life-shortening, and in fact in two of the studies the morbidity and mortality of the animals was actually decreased by PCB treatment. Furthermore, while the tumors are described as malignant, i.e. hepatocellular carcinomas, in none of the three studies did the liver tumors metastasize to other organs even though metastases would be expected if the tumors were malignant. So from these studies it is evident that PCB-treatment does not increase the total cancer risk in these animals, rather it shifts the incidence of the type of tumors observed by significantly decreasing some tumor types while enhancing the incidence of liver tumors. Lastly, PCB mixtures of lesser chlorination, i.e. Aroclor 1254 and Clophen A30 (similar in composition to Aroclor 1242, see table 1.4, page 8, of Brinkman and DeKok, 1980), have been examined in two separate studies and found not to be carcinogenic. Thus, conclusions to be drawn from the rat data, like the mouse data, are specific for the dose and degree of chlorination of the PCB mixture being tested.

Possibly because PCBs produce liver hypertrophy, they enhance the tumorigenesis of certain liver carcinogens if given after the carcinogen in question has had an opportunity to initiate tumors. However, if the PCB exposure precedes or is concomitant with exposure to liver carcinogens the tumorigenic response is typically decreased, probably as a result of an enhanced metabolic detoxification of the carcinogen.

To summarize, the qualitative human relevance of the carcinogenic activity of PCBs based on the animal data is limited. The studies providing some evidence of its carcinogenic activity are specific for the degree of chlorination of the PCB mixture, the total tumor incidence is not increased, and the tumors produced occur only very late in the life of the animal and have no adverse effect on the morbidity or mortality of the animal. There are other considerations that limit concern for the carcinogenic data in rats as well. PCBs are not mutagenic, and the mechanism of tumorigenesis for these compounds therefore would appear to involve an epigenetic mechanism. There is also substantial evidence that the doses used to induce tumors in rats are hepatotoxic, and evidence indicating the neoplasms induced by PCBs are reversible if the exposure is terminated before the animal has been exposed for a considerable portion of the animal's lifespan. All of these findings seriously undermine the human relevance of the animal carcinogenicity data. This is particularly true as the human dosages from past and present human exposures are far lower than those used in the animal studies. Given these considerations, it is concluded that the animal data provides *sufficient evidence* of *limited human relevance* that PCBs of 60% chlorine content (e.g. Aroclor 1260/Clophen A60) are carcinogenic in animals. For PCBs having a chlorine composition of 54% (e.g. Aroclor 1254/Kanechlor 500) there is only *inadequate evidence* of carcinogenicity in animals because the larger and longer study in rats was negative, a finding suggesting that the reversible effects reported in mice may have resulted from a promotion of the substantial background incidence of liver tumors occurring in this species. For the remaining commercial PCB mixtures (i.e. Aroclor 1248, Aroclor 1242, Kanechlor 400, Kanechlor 300, and Clophen A30) there is either *no evidence* or *insufficient evidence* for the carcinogenicity of these mixtures in animals.

Evaluation of Epidemiological Evidence

Evidence for human carcinogenicity would be supplied if the two major epidemiological studies of PCB-exposed workers demonstrated a consistent increase in cancer mortality. This increase should show a positive correlation with exposure and evidence of latency. The evidence for carcinogenicity would be strengthened if the incidence of specific types or sites of neoplasms were consistently elevated. In reviewing the data from these two cohorts, none of these conditions are met.

While Bertazzi et al. (1986), in their study of Italian PCB-exposed workers, found a statistically significant elevation in the rate of cancer mortality among both male and female workers, the much larger study of Brown found no increase in cancer mortality. In the Brown study, there were higher-than-expected incidences of rectal and liver cancer. Evidence for an association between PCB exposure and these malignancy types cannot be considered strong, however, in that: 1) no cases of rectal cancer were observed after the initial report, suggesting that this increased rate was anomalous, 2) the number of cases of liver cancer observed in this study is not appreciably greater than expected when examined without total number of liver neoplasms used by Brown which includes those liver cancers that have metastasized from other organs, 3) in comparison, the study by Bertazzi et al. found only one case of liver cancer and no cases of rectal cancer in their cohort, and 4) the absence of a clear association with latency or relationship with duration of exposure.

The primary sites of neoplasms contributing to the higher-than-expected cancer mortality rates in the Bertazzi studies were located in the digestive system and the hematopoietic and lymphatic systems. These were not increased in the larger cohort reported by Brown. Further, in the Bertazzi cohort, there was no evidence of latency or relationship between cancer mortality and exposure to PCBs. It should also be noted that in the larger of the two subdivisions of their cohort, the female workers, differences in incidences of causes of death presumably unrelated to PCB exposure (viz., increases in accidental death and decreases in deaths from cardiovascular disease) were of similar magnitude as increases in death from malignant tumors. This suggests, at least for this group, that other confounding variables may exist.

In summary, epidemiological evidence for human carcinogenicity of PCBs is at present weak and mostly negative. As such, until larger epidemiological studies can be completed, the data must be considered *inadequate to characterize PCBs as human carcinogens*.

Summary Table

Aroclor 1260/Clophen A60 :

There is *sufficient evidence* of *limited human relevance* for the carcinogenicity of Aroclor 1260 in animals.

The human evidence for carcinogenicity of this compound is *inadequate* .

Aroclor 1254/Kanechlor 500 :

There is *inadequate evidence* for the carcinogenicity of Aroclor 1254 in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

Aroclor 1248/Kanechlor 400 :

There is *no evidence/insufficient evidence* for the carcinogenicity of Aroclor 1248 in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

Clophen A30/Kanechlor 300/Aroclor 1242 :

There is either *no evidence* or *insufficient evidence* for the carcinogenicity of these mixtures in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

BIOLOGICAL DATA RELEVANT TO THE EVALUATION OF CARCINOGENIC RISK TO HUMANS

1.0 Carcinogenicity and Related Studies In Animals

1.1 Studies in Mice

Nagasaki et al. (1972) initially examined the hepatocarcinogenic effects of Japanese brands of PCB fluids by feeding male mice dietary levels of 100 ppm, 250 ppm, and 500 ppm for 32 weeks. As this particular citation represents a short communication, detailed discussions of the experimental design and histopathological examinations of this investigation were lacking. An increased incidence of liver tumors was found in only one of the treatment groups, those mice receiving the 500 ppm diet of Kanechlor 500, where hepatomas were identified in 58% or 7/12 of the animals. The livers from animals in this group also contained nodular areas and many necrotic foci. In sharp contrast to these findings, no hepatomas and none of these histopathologic changes were observed in animals receiving lower doses of Kanechlor 500 or in any of the animals receiving Kanechlor 400 or Kanechlor 300.

The results of the above study also appear to have been reported in two other journals under Ito et al. (1973a&b). As in the previous study it was reported that male mice of the ddY strain were fed diets containing Kanechlor 300, Kanechlor 400 or Kanechlor 500 at dietary levels of 100, 250 or 500 ppm for 32 weeks. PCBs significantly increased the liver weights of the animals, and at the highest dose of Kanechlor 500 the liver:body weight ratio had increased 3-fold with histopathological examination of the livers revealing a focal hypertrophy in the centrilobular hepatocytes of non-neoplastic areas. Another change observed in all PCB treatment groups, except those receiving 500 ppm of either Kanechlor 500 or 400, was a marked amyloid degeneration of the liver in the space between the sinus endothelium and the hepatocytes. For some reason, the liver tumors in these more recent publications (Ito et al, 1973a&b) have been reclassified, and the tumors reported as hepatomas in the previous communication (Nagaski et al., 1972) are now listed as carcinomas (Ito et al., 1973 a&b). That is, in the group receiving 500 ppm of Kanechlor 500 (originally described as having hepatomas in 7/12 animals) nodular hyperplasia was found in 7/12 animals (58.3%) while 5/12 of the livers (41.7%) now have well-differentiated hepatocellular carcinomas. The carcinoma cells were reported to be comprised of irregularly shaped cells with pyknotic nuclei and occasionally had mitotic nuclei. The reason for changing the classification of these neoplasms from hepatoma to carcinoma is not provided. Of some interest to discussions of the carcinogenic potential of PCBs is the fact that all of the other doses of all three Kanechlors tested failed to produce even nodular hyperplasia. Thus, the tumorigenicity reported was quite specific for the dose and the extent of chlorination of the Kanechlor being tested.

Kimbrough and Linder (1974) examined the effects of PCBs in

ce and obtained results consistent with those found for the lesser-chlorinated Kanechlors of the preceding study. Groups of 50 male Balb/cJ mice were fed 300 ppm Aroclor 1254 for either 11 months or for 6 months followed by a 5 month recovery period. The authors pointed out that this dietary level was some 2,500 times the Food and Drug Administration's estimate of the average human daily PCB intake from food during the early 1970's. About one-half of the animals in each PCB treatment group died during the first 4 months, but this problem seems to have been unrelated to the PCB treatment as control animals suffered a similar incidence of mortality. Of the 22 mice surviving the 11 month feeding study, all animals had hepatomegaly with the liver representing approximately 25% of their body weight in comparison to the 5.8% liver:body weight ratio measured for the control animals. Adenofibrosis (i.e. cholangiofibrosis) was observed in all 22 livers taken from mice fed PCBs for 11 months, however 13/68 livers taken from control animals had occasional, small areas of necrosis and fibrosis. Of the livers taken from animals receiving 300 ppm Aroclor 1254 for 11 months, the nuclei were enlarged, hyperchromatic and atypical. The cytoplasm was either smooth or vacuolated, and the Kupffer cells contained a brown pigment. Some of these livers had extensive necrosis and fibrosis. In each of the livers, several areas of hepatocytes had been replaced by proliferating epithelial cells which formed ducts and often produced mucus. In addition to these histological changes, 10 hepatomas were found in 9 of these livers. The tumors were described as well circumscribed and surrounded by compressed parenchyma or strands of fibrosis. In the 24 surviving mice fed PCBs for only 6 months followed by a 5 month recovery period, only one liver contained a hepatoma. Yet, liver fibrosis was observed in two-thirds of these animals and hepatocellular necrosis was evident in most of the livers.

1.2 Studies in Rats

Kimura and Baba (1973) exposed 10 male and 10 female Donryu rats to a variable dietary level of Kanechlor 400 for 400 days. The diet initially contained 38.5 ppm and was fed to the animals for 4 weeks; the dietary level was then doubled and provided for the following 8 weeks; the initial dietary level was then increased 4-fold and provided for 3 weeks; it was then increased 8-fold and fed to the animals for another 3 weeks; finally it was increased to 16 times the initial level and fed to the animals for 3 more weeks. This last increase in the dietary levels of PCBs (a level that was approximately 616 ppm) was found to be too toxic and caused a considerable weight loss in the animals. In response to the toxicity observed at this dose, the dietary level was reduced to 462 ppm for the remaining 32 weeks of the study. Further complicating interpretations of this study is the fact that animals died or were sacrificed at various times throughout the experiment, therefore the total PCB dose each animal received may differ. In general, the total amount ingested was thought to be 1300-1800 mg for the group of male animals and 1100-1500 mg for the female animals. Microscopically the livers of all of the treated animals

contained fatty degenerative changes, and while 6/10 of the livers from female animals had adenomatous nodules, none of the livers of the male animals contained such nodules. However,the liver nodules observed in the female animals do not appear to be related to the PCB treatment as 2/5 (40%) of the livers from the control female animals also contained adenomatous nodules.

In a second study Kimura et al. (1976) fed 12 female Donryu rats diets containing 400 ppm Kanechlor 400 for 6 months. The estimated dose corresponded to a total of 531 mg of PCBs during this period. Eight of the 12 animals were then sacrificed 590 days after the feeding began. None of these animals developed hepatocellular carcinoma, and 9/12 of the livers were normal in appearance, suggesting that the degenerative changes observed in the previous study are reversible.

Ito et al. (1974) fed male Wistar rats Kanechlor 300, Kanechlor 400 or Kanechlor 500 at dietary levels of 100, 500 or 1,000 ppm for up to 52 weeks. No hepatocellular carcinoma was found in the livers of any of the treated rats. The highest dose of all three Kanechlors did produce a cholangiofibrosis of the liver, but this effect was not observed at the lower doses with any of the Kanechlors. Nodular hyperplasia was observed in 30-40% of the rats exposed to the two highest doses of Kanechlor 500 (i.e. doses of 500 ppm and 1.000 ppm) and in animals receiving a diet containing 1,000 ppm of Kanechlor 400. Oval cell proliferation and proliferation of the bile duct cells were observed in all treatment groups. Hypertrophy of the centrilobular cells was also evident in animals receiving the highest dose of the two most heavily chlorinated PCB mixtures. Fatty changes and fibrosis were also observed in the livers of animals of several of the treatment groups. The fatty changes, hypertrophy and fibrosis of the liver all tended to be present and correlate with the observation of nodular hyperplasia, suggesting that these changes may have been contributory factors.

Kimbrough et al. (1975) published the first major positive study demonstrating that Aroclor 1260 can produce hepatocellular carcinoma in the rat. In this study 200 female Sherman strain rats were fed Aroclor 1260 at a dietary level of 100 ppm for approximately 21 months. There was a statistically significant 6-7% decline in the weight gain of the animals exposed to PCBs in this study suggesting that the dose used approximated the maximally-tolerated dose. The incidence of the histopathological findings from this study are summarized in Table 1. The most consistent histopathologic difference in the PCB treatment group was the finding of hyperplastic or neoplastic nodules in 144/184 (78%) of the livers. More importantly, however, was the finding of hepatocellular carcinoma in 26/184 (14%) of the PCB-treated animals. The tumors were well-differentiated neoplasms of the trabecular type, except in three of the animals which had tumors with a glandular, papillary pattern. Foci of coagulative necrosis were occasionally observed in the cancerous areas, but there was no fibrosis or other evidence of chronic degenerative changes. Tumors

in areas other than the liver were not listed as significantly different, however, in some cases there was a substantial decrease in the tumor incidence of other tissues, e.g.

TABLE 1

Results of Carcinogenesis Bioassay of Aroclor 1260

Organ/Tissue	Tumor Type	Tumor Incidence	
		Control Animals	Aroclor 1260 Animals
Liver	Hepatocellular carcinoma	1/173	26/184
	Neoplastic nodules	0/173	144/184
	Areas of cytoplasmic alteration	28/173	182/184
Thyroid gland	Parafollicular cell tumor	37/160	18/166
Pituitary gland	Chromophobe adenoma	41/153	28/139
Mammary gland	Fibroadenoma	17/173	13/184
Ovary	Granulosa theca cell tumor	5/149	0/163

Adapted, in part, from Kimbrough et al. (1975)

as in the case of parafollicular cell tumors of the thyroid. The total incidence of extra-hepatic tumors in the Kimbrough et al. (1975) study was 134/173 (74%) in control animals versus 110/183 (60%) in Aroclor 1260 treated animals. So while PCBs had significantly increased the liver carcinoma incidence, the total tumor incidence (78% in control animals -vs- 74% in Aroclor animals) in PCB-treated animals was actually slightly less than the incidence for control animals. The PCB treatment was also not life-shortening; on the contrary, about twice as many control animals had died for various reasons before the experiment was terminated at 23 months.

Calandra (1976) at the 1975 PCB conference sponsored by the EPA reported the findings of several chronic studies performed by a commercial laboratory for Monsanto. While these studies have never been published the results were recently reviewed by Levinskas (1981). In this study 1,000 rats were divided into ten groups of 100 animals (50 of each sex) and nine of these groups were exposed to Aroclors 1242, 1254 or 1260 at dietary levels of 1, 10 or 100 ppm. Apparently five animals of each sex were sacrificed at 3, 6 and 12 months with about 35 animals to be killed at the end of the two year study (EPA, 1980). In the animals sacrificed early, only

one nodular hyperplasia was observed and it was in the group fed 100 ppm of Aroclor 1260 for 12 months. Mortality in this study was high and approximately one-third of the 105 animals anticipated to be exposed for two years at each dietary level died. Hepatomas were observed in 7/25 livers from animals fed 100 ppm Aroclor 1260, in 4/26 fed Aroclor 1254, 3/19 fed Aroclor 1242, and only 1/168 animals receiving the 1-10 ppm diets. Nodular hyperplasia was twice as prevalent as hepatomas in the high dose animals, particularly in the Aroclor 1254 group.

In 1978 the National Cancer Institute examined the carcinogenic potential of Aroclor 1254 (NCI, 1978). Groups of 24 male and 24 female Wistar rats were fed Aroclor 1254 at dietary levels of 25, 50 or 100 ppm for 105 weeks. Clinical signs of toxicity including hair loss, facial edema and cyanosis occurred by week 72 in the high dose animals and the mean body weights were roughly only 2/3-3/4 that of their respective controls. This decrease in body weight exceeds the no more than 10% weight loss guideline for the estimated maximally tolerated dose that is part of the NCI guideline for cancer bioassays (NCI, 1979). Several histopathologic changes occurred in the livers of animals receiving PCBs that appeared to be related to the PCB treatment, particularly the incidence of hyperplastic nodules and adenomas. Male animals had one hepatocellular carcinoma in the 50 ppm group and only 2 in the 100 ppm group. Although the incidence of these tumors was not significant, the occurrence of proliferative lesions did appear to be dose related. In reviewing this bioassay, the Data Evaluation/ Risk Assessment subgroup of the Clearinghouse on Environmental Carcinogens responsible for providing peer review of NCI studies concluded the following :

> "It is concluded that, under the conditions of the bioassay, Aroclor 1254 was <u>not carcinogenic</u> in Fischer 344 rats; however, a high incidence of hepatocellular proliferative lesions in both male and female rats were related to treatment. In addition, the carcinomas of the gastrointestinal tract may be associated with treatment in both males and females. Based on the liver proliferative lesions in the treated rats and published reports, it is suggested that Aroclor 1254 may be a tissue promoter."

Morgan et al. (1981) have taken the same tissue sections that originated in the NCI bioassay (NCI, 1978), stained the stomach sections for alkaline phosphatase, and then re-sectioned these tissues for histological evaluation in conjunction with those provided in the NCI study itself. The final incidence of alkaline phosphatase rich areas was 6.4% in controls, 10.4% in animals fed 25 ppm Aroclor 1254, 16.7% in the 50 ppm group, and 35.4% in the 100 ppm group. These changes were most often noted in the pyloric region of the stomach and duodenum (88% of the lesions were found in these areas), suggesting a toxicity specific to the cells of these areas. Gastric adenocarcinomas comprised six of the 33 total

lesions identified in these slices. Three were found in tissues from the 50 ppm treatment group and two in the animal group fed 100 ppm. Thus, the increased incidence of this tumor was not related to the dose of PCBs. The remaining 27/33 lesions identified in this study were described as intestinal metaplasia. The authors concluded that the actual number of lesions they believed should have been observed in the G.I. tract tissue sections of the NCI study was twice the number of lesions reported in the original NCI study. Further, and on the basis of their findings, the authors of this paper concluded that chronic oral Aroclor 1254 exposure may lead to the induction of intestinal metaplasia, and possibly to adenocarcinoma of the glandular stomach of the Fischer 344 rat (Morgan et al., 1981).

Ward (1985) has also published a review of the slides originating from the NCI bioassay. In addition to the aforementioned dose-related depression of body weight, Ward (1985) also discusses, in some detail, the substantial decrease in animal survival that occurred in this study. While the survival rate in control animals and in the treatment group receiving 25 ppm was 92% and 83%, respectively, only 58% of the animals receiving the 50 ppm diet and 46% of the animals fed diets containing 100 ppm survived to the end of the bioassay. Focal hyperplasia was of the eosinophilic type and was only observed in PCB-treated animals. If compression was found on two sides of the neoplasm, Ward diagnosed the lesion as hepatocellular adenoma. A total of 13 eosinophilic, basophilic or vacuolated adenomas were identified. All of these occurred, with one exception, in those animals fed the two higher dietary levels of Aroclor 1254 and the occurrence of adenomas was slightly greater in the male animals (8/13). Similar to the findings of the NCI (1979) report, only two liver carcinomas were identified, both occurred in male animals receiving the 100 ppm diet. Ward (1985) also reported that Aroclor 1254 increased the incidence of intestinal metaplasia and gastric adenocarcinoma. As in his earlier report with Morgan and Hartman (Morgan et al.,1981), the change in adenocarcinoma was neither significant nor dose-related. A significant increase in intestinal metaplasia was only observed in the 100 ppm dose group. Thus, the Ward (1985) study is still consistent with the previous NCI (1979) bioassay. No statistically significant increase in liver cancer or cancer of other tissues was observed, but the 100 ppm dose does lead to significant intestinal metaplasia. Ward mentions the fact that the liver lesions he observed were predominantly of the eosinophilic type rather than the basophilic type generally observed in the control animals. Based on these changes Ward proposes the idea that these data may suggest that PCBs are capable of initiating liver tumors rather than promoting the background tumor incidence. Yet, in contradiction of his suggestion, Ward also makes note of the fact that inducing agents like PCBs and phenobarbital cause a proliferation of the smooth endoplasmic reticulum (SER) of the liver. As a proliferation of the SER gives rise to an eosinophilic appearance of the cytoplasm, the liver hypertrophy and induction of microsomal enzymes associated with PCB exposure provides an obvious explanation for the basophilic to eosinophilic change Ward noted in

the cellular appearance of the liver tumors of PCB-treated animals.

Schaeffer et al. (1984) used a total of 432 weanling Wistar rats to examine the effects produced by chronically feeding rats Clophen A60 (equivalent to Aroclor 1260) or Clophen A30 (similar in composition to Aroclor 1242; Brinkman and DeKok, 1980). The study consisted of three groups. Group 1, a control group of 139 animals receiving the normal diet, Group 2 with 152 animals receiving a diet containing 100 ppm of Clophen A30, and Group 3 which consisted of 141 animals fed a diet containing 100 ppm of Clophen A60. After day 801 animals were randomly selected and killed, and the experiment was terminated on day 832. The Clophens used in this study were reported to be free of any chlorinated dibenzofuran contamination, but the level of detection for this analysis was not specified. In those animals necropsied prior to day 800, hepatocellular carcinomas were only identified in the PCB treatment groups, one in Group 2 and a total of 9 were observed in Group 3. This latter number was statistically significant for the Clophen A60 treatment, but represented a liver cancer incidence of only 7% for the entire group. In contrast, the incidence of thymoma was significantly reduced by PCB treatment declining from 12% in the control group to 3-4% in the treatment groups. Likewise the total number of the remaining types of neoplasms was significantly reduced by the PCB treatment, with Clophen A60 causing the greatest reduction (from 52 in controls down to 18 in the Clophen A60 group). The final results of this study are shown below in Table 2. The incidence of hepatocellular carcinoma was significantly increased only in those animals receiving the Clophen A60. Thus, the results of this study were consistent with the previous rat studies, i.e. Aroclor 1260, or its equivalent, was reported to have induced hepatocellular carcinoma in rats while a lesser chlorinated PCB mixture was not carcinogenic.

Table 2

Frequency of Hepatocellular Alterations Induced by Chronic Feeding Studies with Clophen A30 and Clophen A60

	# of Foci	Neoplastic Nodules	Hepatocellular carcinoma
Controls (group 1)	6/131 (4.5%)	5/131 (3.8%)	1/131 (0.8%)
Clophen A30 (group 2)	63/130* (48%)	38/130* (29%)	4/130 (3%)
Clophen A60 (group 3)	3/126 (2.4%)	63/126* (50%)	61/126* (48%)

* denotes a significant difference from the control value ($P<0.05$)

In a letter to the editor, Young (1985) comments on several additional, interesting and important aspects of the Schaeffer et al. (1984) study that were not noted by the authors of this paper. Young's analysis of this data focused on the effects of PCB exposure on tumor incidence in liver, on tumor incidence in extra-hepatic tissues, and on mortality. The tables generated by Young (1985) are provided below in Table 3.

Table 3

The Incidence of Hepatocellular Carcinoma, Other Neoplastic Lesions, and Mortality by Time Period

Time Interval (days)	Treatment Group: Control	Clophen A30	Clophen A60
A. Hepatocellular carcinoma			
301-400	0/137	0/150	0/135
401-700	0/111	0/122	0/115
701-800	0/92	1/107	9/85*
B. Other Neoplasms			
301-400	0/137	1/150	2/150
401-700	32/111	11/122*	9/115*
701-800	30/92	15/107*	7/85*
C. Incidence and Percentage(%) of Mortality			
101-400	2/139 (1.4%)	2/152 (1.3%)	3/141(2.1%)
401-700	45/137(32.8%)	43/150(28.7%)	23/138(16.7%)
701-800	39/92 (42.4%)	20/107(18.7%)	30/85 (35.3%)
1-800	86/139(61.9%)	65/152(42.8%)*	56/141(39.7%)*

* Different from control value ($P<0.05$), χ^2

Young points out that the Schaeffer et al. (1984) study actually demonstrated three things. These are : 1) that a significant increase in hepatocellular carcinoma occurred only in the animals receiving Clophen A60, 2) that PCB treatment resulted in a significant decrease in other neoplastic lesions, and 3) that PCB treatment significantly increased the chances of survival of the animals. Given these findings, Young states that it is difficult to conclude, given the balance of the data, whether or not the PCB treatment was in fact detrimental to the rats in light of the fact that the PCB treatment significantly enhanced the rate of survival

and significantly decreased the total tumor load of the exposed rats. In essence, he questions the human relevance of tumors which occur only very late in the life of the animal, are not life-shortening and do not metastasize to other organs of the body. Thus, to quote Young :

> "If the purpose of long-term studies is to extrapolate to humans, then one finds it difficult to infer dire consequences to humans when the treatment is beneficial in the model system. Is the model only useful for inferring bad events? The model should be equally valid for detrimental and beneficial effects."

Lastly, the analysis of Young also calls to question a suggestion made by Schaeffer et al. (1984), which was that the decrease in the incidence of thymoma might be caused by immunosuppressive effects of PCBs. While PCBs cause thymic atrophy at certain doses, any proposed immunosuppressive effect cannot be considered to have a significant clinical impact when the treated animals did not ultimately suffer a greater incidence of morbidity or mortality from either infectious diseases or the cancer induced by this treatment.

The last major rat study reported is that of Norback and Weltman (1985). These investigators fed 70 male and 70 female Sprague-Dawley rats a diet containing 100 ppm Aroclor 1260 for 16 months followed by a reduction to 50 ppm for the next 8 months. The animals were then fed a control diet for the remaining 5 months of their lives. All results were compared to a control group which initially consisted of 126 animals, 63 of either sex. At various time points throughout this study two control animals of each sex and three PCB-treated animals of each sex (10 animals in total) were anesthetized with ether and the medial left lobe of the liver of each animal was surgically removed. These tissue samples were taken at 1, 3, 6, 9, 12, 15, and 18 months. At 24 months a similar group was killed and at the end of 29 months all remaining animals were sacrificed. The induction of liver hypertrophy in the centrilobular area of the lobule was evident at the first observation period made one month after the PCB diet was initiated. By the 18th month the liver:body weight ratio had increased from 4% to 12% in the female animals. Macroscopically these investigators noted evidence characteristic of neoplastic nodules near the capsular surface, hepatocellular carcinomas and adenofibrosis. In the PCB-exposed group, the observed lesions appeared in the following sequence : centrilobular hypertrophy at 1 month, foci of cells appeared at 3 months, foci of altered cells in the centrilobular and midzonal regions at 6-9 months, neoplastic nodules appeared at 12 months, trabecular carcinoma was observed after 15 months and adenocarcinoma at 24 months. Simple cystic cholangioma and adenofibrosis appeared in animals 18-23 months after the exposure began. There was no evidence of metastases to the lungs. All trabecular carcinomas had cell arrangements with a glandular, ductal or cystic pattern and all adenocarcinomas had some elements of the trabecular pattern of growth. The lumens of

...he adenocarcinomas were the apparent result of cellular necrosis.

The incidence of tumors in animals 18 months or older are presented below in Table 4. It should be noted that 7-8 animals

Table 4

Incidence of Hepatocellular Neoplasms

	Incidence or % Tumors Observed (The actual number of animals with tumors)		
	Male	Female	Total
A. Control Animals	(N=32)	(N=49)	(N=81)
Trabecular carcinoma	0% (0/32)	0% (0/49)	0% (0/81)
Adenocarcinoma	0% (0/32)	0% (0/49)	0% (0/81)
Number negative	100% (32/32)	98% (48/49)	99% (80/81)
B. Aroclor 1260 Animals	(N=46) †	(N=47) ††	(N=93)
Trabecular carcinoma	4% (2/46)	40% (19/47)	23% (21/93)
Adenocarcinoma*	0% (0/46)	51% (24/47)	26% (24/93)
Neoplastic nodule only	11% (5/46)	4% (2/47)	8% (7/93)
Number negative	85% (39/46)	4% (2/47)	44% (41/93)

† The total number includes 8 animals that had received a partial hepatectomy during the first 18 months.
†† The total number includes 7 animals that had received a partial hepatectomy during the first 18 months.
* Animals with both trabecular carcinoma and adenocarcinoma were placed only in the adenocarcinoma group.

sacrificed after 18 months, i.e. at least 15% of the group of animals in which late developing tumors were observed, had ...ne a partial hepatectomy during the first 18 months. The ... of this cannot be determined from this experiment, but partial hepatectomy has been used as a promotional stimulus to increase the incidence of liver tumors induced by other carcinogens. Therefore, it is unfortunate that the authors did not note or describe the possible influence that this might have had on the final tumor incidence measured. Another important factor to consider, and one which is not readily apparent from Table 4, is that almost all of the tumors reported in this study were very late-developing tumors. In Table 1 of their paper, only 4 trabecular carcinomas and only 2 adenocarcinomas had developed between the 18- and 24-month sacrifices. Thus, 35/41 or some 85% of the liver tumors observed in this study developed in the last 25-29 month period of the study. These tumors had not metastasized to other organs, and none appear to have been life-shortening. Concerning this last aspect, unfortunately no information is given concerning the cause of death for any animals dying early, or concerning the number of animals lost. But from the data supplied

t would appear that a number of early deaths occurred only in the group of male control animals. These same observations were also noted by the authors, who stated in the discussion of this paper :

> " Although the tumors met the morphological criteria for malignancy, their biologic behavior was relatively unaggressive. The neoplasms did not metastasize to distant organs nor invade blood vessels. Mortality of the animals was not increased. The lack of greater morbidity or mortality is likely due to slow progression of the neoplastic process and late appearance and slow growth of the hepatocellular carcinoma."

The authors further noted that it remains to be established whether PCBs have an initiating effect or whether the neoplasms observed result from the promotion of a background incidence of initiated cells.

1.3 Studies In Other Species

Calandra (1976) reported, in summary form, a carcinogenicity bioassay performed in dogs. Groups containing 4 male and 4 female dogs were fed 1, 10 or 100 ppm of Aroclors 1242, 1254 or 1260 for two years. While this exposure interval is for a period of time that is considerably less than the lifetime of the species being tested, no remarkable findings were reported.

1.4 The Effects of PCBs on Other Liver Carcinogens

PCBs have been tested in numerous studies to determine whether they alter the carcinogenicity of other chemicals. The evidence obtained from such studies is sometimes inconclusive, and in several instances similar studies have produced conflicting results. The results of have been summarized in the following paragraphs and in table 5 located on page 13 :

- A number of studies indicate that PCBs administered after a carcinogenic dose of a liver carcinogen enhance the incidence of tumors. Nishizumi (1976) found that Kanechlor 500 promotes liver tumors initiated by diethylnitrosamine. Kimura et al. (1976) demonstrated that Kanechlor 400 can promote the liver cancer induced by 3'-methyl-4-dimethylaminoazobenzene (MeDAB). Ito et al. (1978) initiated liver tumors with N-2-fluorenylacetamide and then increased tumor incidence by feeding the rats diets containing 1,000 ppm of PCBs. Preston et al. (1981) have reported that Aroclor 1254 with or without polychlorodibenzofuran contaminants can promote the liver cancer initiated by diethylnitrosamine. Osterle and Deml (1984) demonstrated that this effect can be produced in weanling rats, although the immature animal is less sensitive. Deml et al. (1983) have also found that an initial PCB pretreatment to increase

oxidative metabolism, followed later by PCB administration for its promotional effects, increased the number of rat liver preneoplastic islands. Similarly, Pereira et al. (1982) demonstrated that Aroclor 1254 promoted the number of diethylnitrosamine-induced enzyme-altered foci in rat livers. Recent work by Deml and Oesterle (1987) has shown that while PCBs are capable of promoting the liver tumors initiated by diethylnitrosamine, the threshold for this effect is about 1 mg/kg/day and correlates strongly to the induction of MFO activity and subsequent liver hypertrophy that the induction of drug metabolism normally produces (Greim et al., 1985).

- In contrast to the above studies, Gans and Pitauro (1986) reported Aroclor 1254 increased the incidence of liver scarring in mice fed diethylnitrosamine, but PCB treatment did not enhance the formation of liver nodules. Arai et al. (1983) also reported that PCBs administered to rats following an initial exposure to dimethylnitrosamine increased the incidence of liver tumors, however, this same dosage regimen decreased the incidence of kidney tumors produced by DEN.

- Conflicting results have been reported in trout where the effect of PCBs is dependent upon the time of PCB administration as well as the carcinogen it is administered with. For example, Hendricks et al. (1980) found that PCBs did not promote the incidence of liver tumors induced by aflatoxin B_1 in trout, even though the trout were fed a diet containing PCBs for 12 months after the aflatoxin B_1 exposure. However, other studies by Shelton et al. (1984a) have demonstrated that PCBs exert an inhibitory effect on the incidence of liver tumors initiated by aflatoxin B1 when PCBs are co-administered with this liver carcinogen. Alternatively, trout fed diets containing 1100 ppm diethylnitrosamine + 100 ppm of either Aroclor 1242 or Aroclor 1254 had a greater incidence of liver cancer than did trout fed diets that only contained 1100 ppm DEN (Shelton et al., 1984b).

- Loury and Byard (1983) observed that Aroclor 1254 enhanced the DNA repair response of amino acid pyrolysate mutagens in primary hepatocyte culture when the PCBs were given three days prior to cell isolation. Thus, it appears that these effects are related to a PCB enhancement of oxidative metabolism. In contrast, Nesnow et al. (1981) found that Aroclor 1254 did not enhance the benzo(a)pyrene-mediated transformation of C310T1/2CL8 mouse embryo fibroblasts even though a number of other chemicals known to induce drug metabolism were positive in this test.

- PCB exposure <u>in utero</u> and during nursing protected

Wistar rats and CD-1 mice from diethylnitrosamine-induced liver tumors (Nishizumi, 1980, Anderson et al., 1983). Likewise Kimura et al. (1976) found that PCBs given before or during MeDAB exposure decreased the incidence of MeDAB-induced liver cancer. Similarly, Makiura et al. (1974) have shown that the co-administration of PCBs with the liver carcinogens MeDAB, N-2-fluorenylacetamide and diethylnitrosamine decreases tumor incidence. Trout fed diets containing PCBs prior to their exposure to aflatoxin had a reduced frequency of liver tumors induced by the aflatoxin (Hendricks et al. 1977). This effect is perhaps explained by Stott and Sinnhuber (1978) who demonstrated that PCBs reduced the bioactivating ability of trout microsomal fractions used to test the mutagenicity of aflatoxin B_1.

- In dermal initiation-promotion assays PCBs have been found to diminish the initiating activity of certain skin carcinogens and to be devoid of any promoting activity (DiGiovanni et al., 1977; Berry et al., 1978). Similarly Hayes et al. (1985) examined the effects of PCBs on the proliferating hepatocytes in livers of young animals or adult animals after partial hepatectomy. They concluded that short-term PCB exposures do not have an initiating action in an *in vivo* assay that detects both hepatic and extra-hepatic initiating carcinogens.

- In contrast to its reportedly adverse effects on the immune system, Kerklivet and Kimeldorf (1977a&b) demonstrated that PCBs administered either in the diet or by injection reduced the size of Walker 256 carcinosarcomas transplanted to rats and increased the lifespan of the PCB-treated animals.

Table 5

Summary of PCB Interactions With Other Carcinogens

Study	Carcinogen	Results
A. PCBs Administered After the Carcinogen :		
Nishizumi (1976)	diethylnitrosamine	promotion (liver tumors)
Kimura et al., (1976)	Me-DAB	promotion (liver tumors)
...to et al., (1978)	N-2-fluorenylacetamide	promotion (liver tumors)

Hendricks et al., (1980)	aflatoxin B1	no effect
Preston et al., (1981)	diethylnitrosamine	promotion (liver tumors)
Pereira et al., (1982)	diethylnitrosamine	promotion (liver tumors)
Deml et al., (1983)	benzo(a)pyrene	promotion (liver tumors)
Arai et al., (1983)	dimethylnitrosamine	promotion (liver tumors) inhibition (kidney tumors)
Gans & Pitauro, (1986)	diethylnitrosamine	liver scarring no effect (liver nodules)
Deml & Oesterle, (1987)	diethylnitrosamine	promotion (liver tumors) threshold dose (1 mg/kg/day)

B. PCBs Administered Before or With the Carcinogen :

Makiura et al., (1974)	Me-DAB	less liver tumors
	N-2-fluorenylacetamide	less liver tumors
	diethylnitrosamine	less liver tumors
Kimura et al., (1976)	Me-DAB	less liver tumors
Hendricks et al., (1977)	aflatoxin B1	less liver tumors
Nishizumi (1980)	diethylnitrosamine	less liver tumors
Anderson et al., (1983	diethylnitrosamine	less liver and lung tumors
Shelton et al., (1984a)	aflatoxin B1	less liver tumors
Shelton et al., (1984b)	dimethylnitrosamine	promotion

Me-DAB = 3'-methyl-4-dimethylaminoazobenzene

It is concluded from these studies that the liver hypertrophy induced by PCBs can and does enhance the carcinogenic response of initiating hepatocarcinogens if the PCBs are given some time after the liver cells have been initiated by the other chemical.

...ever, this effect has also been demonstrated for phenobarbital and is no doubt also true for most other liver enzyme inducing chemicals that produce a hypertrophy or hyperplasia of the liver. Additionally, recent work by Deml and Oesterle (1987) has shown that the promotional effects of PCBs has a threshold, as one might expect, and this threshold appears to be well above the dose commonly encountered by most persons. In contrast to its promotional effects on liver carcinogens known to be initiators, PCBs also act as anticarcinogens when administered prior to the exposure of an initiating carcinogen. This effect is most probably the result of PCB induction of liver metabolism which then acts to decrease the carcinogenic effects of a number of chemicals by increasing their degradation and elimination.

2.0 Epidemiological Studies of PCB Exposure and Cancer Mortality

Published epidemiological studies addressing the carcinogenic potential of PCBs have been confined to two cohorts, both consisting of employees of plants where PCBs were used in the manufacture of capacitors.[Note : Bahn et al. (1976), in a letter to the editor, reported an analysis of eight deaths among 92 research and development workers at a refinery, and suggested an association between PCB exposure and incidence of malignant ...lanoma. The extremely limited size of their cohort, the inability ...adequately assess exposure of the cohort to other, potentially carcinogenic substances or other known risk factors, and the inconsistency of their observations with the results from the two larger cohorts, precludes any utility of this report in the assessment of the human carcinogenicity of PCBs.]

Brown and Jones (1981) have published a retrospective cohort mortality study of employees from two plants. Minimal exposure period to PCBs was three months, and the types of PCBs used were Aroclors 1016, 1242, and 1254. Workers who had potential exposure to trichloroethylene, which was also used in both plants, were excluded from the cohort. Of the 2567 employee cohort, 163 were known to be deceased and the vital status of 55 (2%) was unknown.

Higher-than-expected incidences of cancer in the rectum (primarily among females in Plant #2) and liver were noted, although the differences were not statistically significant. The overall incidence of malignancies observed was slightly less than expected. None of the causes of death showed a clear association with latency, and no relationship was observed between the duration of employment in jobs involving PCB exposure and risk of mortality due to cancer. The causes of mortality and incidences of specific malignant neoplasms are summarized in Table 6.

The study of mortality in this cohort has recently been updated, with the number of deceased increasing from 163 to 295

(Brown, unpublished data). There were no additional deaths from cancer of the rectum which brought the observed incidence of this form of neoplasm closer to the expected value. The number of observed malignancies of the liver was significantly greater than expected. However, a number of these malignant neoplasms (3/5) appear to have been the result of metastasis from other sites. Thus, like the rectal cancer the standard mortality rate (SMR) for liver cancer originating in the liver has been substantially lowered by this larger cohort. A higher-than-expected incidence of neoplasms was not noted for any other tissue. The overall incidence of malignancies observed was less than expected. There was no apparent relationship between duration of employment in "PCB-exposed" jobs or total employment and cancer mortality of any type.

Table 6

Major Causes of Death for Plant Workers Exposed to Aroclor 1016, 1242, and 1254

Causes of Death	Observed/Expected	Standard Mortality Rate
	All Causes of Mortality	
All malignant neoplasms	39/43.79	(89)
Diseases of nervous system	11/12.55	(88)
Diseases of circulatory system	60/62.93	(95)
Accidents	13/18.29	(71)
All other causes	40/44.79	(80)
Total mortality for all causes	163/182.35	(89)
	Malignant Neoplasms	
Cancer of stomach	1/1.66	(60)
Intestine	4/4.03	(99)
Rectum	4/1.19	(336)
Liver	3/1.07	(280)
Pancreas	1/1.90	(53)
Respiratory	7/7.98	(88)
Breast	7/6.84	(102)
Lymphatic	2/4.34	(46)
Other	10/14.78	(68)

Source: Brown and Jones (1981)

The first study of the second major cohort was reported by Bertazzi et al. in 1981. Inclusion in the cohort was limited to workers in the production department who had been employed for at least six months. There were a total of 1310 workers included in

the study, and exposure was primarily to Aroclor 1254 and Pyralene 476. While some members of the cohort were also exposed to trichloroethylene, alkylbenzene, and epoxy resins, the number was stated to be few. A total of 27 deaths were observed in the cohort between 1954 and 1978. The overall incidence of cancer observed was significantly higher than expected (14 versus 5.65 expected), contributed primarily by elevated (though not statistically significant) incidences of neoplasms of the digestive organs and lymphatic and hematopoietic systems.

Bertazzi et al. (1986) have recently published an update of their study of PCB-exposed capacitor workers with a modified cohort. Non-production workers at the plant were added to the cohort, and the minimum period of employment was reduced from six months to one week. The size of the cohort was consequently increased to 2100, and the number of deceased to 64. The results from male and female workers were analyzed separately, and are summarized below in Tables 7 and 8.

Table 7

Mortality From Selected Causes of Male Workers Exposed to PCB

		Reference Cohort			
		National Mortality Estimates		Local Mortality Estimates	
Cause of death	Observed	Expected	SMR	Expected	SMR
All causes	30	27.8	108	29.8	101
Malignant tumors	14	5.5	253[a]	7.6	183[b]
Cancer of G.I. tract	6	1.7	346[c]	2.2	274[d]
Lung cancer	3	1.2	250	1.6	187
Hematologic neoplasms	3	0.8	375	1.1	263
Cardiovascular disease	8	7.9	101	9.4	95
Accidents	6	6.8	88	5.8	103

Confidence limits (95%) [a]= 144-415 [c]= 141-721
[b]= 104-300 [d]= 112-572

For males, mortality from cancer was significantly greater than expected (14 versus 5.5 expected). Among specific tissues, the observed incidence of neoplasms of the GI tract was significantly greater than expected, and hematologic neoplasms were also greater

than expected but not statistically significant. Overall mortality was only slightly greater than expected. Among females, higher-than-expected incidences of overall mortality, mortality from cancer, and incidence of hematologic neoplasms were observed. These differences were of statistical significance when the cohort was compared to local mortality rates but not national mortality rates. While neoplasms were grouped by organ-system for statistical analysis, specific tumor sites were noted. Among both male and female members of the cohort, there was one case of liver cancer and no cases of rectal cancer. There was no apparent association between duration of exposure, latency, and year of first exposure for any of the causes of mortality.

Table 8

Mortality From Selected Causes of Female Workers Exposed to PCB

		Reference Cohort			
		National Mortality Estimates		Local Mortality Estimates	
Cause of death	Observed	Expected	SMR	Expected	SMR
All causes	34	25.8	132	16.5	206[a]
Malignant tumors	12	7.7	156	5.3	226[b]
Hematologic neoplasms	4	1.5	266	1.1	377[c]
Cardiovascular disease	2	4.7	42	3.0	66
Accidents	9	4.0	225	4.0	225

Confidence limits (95%) [a]= 145-285 [c]= 115-877
[b]= 123-385

From Bertazzi et al., 1986

3.0 Other Relevant Human Data

3.1 Health Effects Information from Accidental Ingestion

In 1968, the ingestion of PCB-contaminated rice oil in Japan resulted in an outbreak of chloracne and other symptoms which were later termed "Yusho". A similar large-scale incident of ingestion of contaminated rice oil occurred approximately ten years later in Taiwan with similar symptoms. The most common symptoms observed in rice oil poisoning victims included increased discharge from the eyes, swelling of the eyelids, accompanying visual disturbances, fatigue and malaise, headache, and symptoms suggestive of peripheral neuropathies (numbness of the limbs and pruritis) (Higuchi, 1976; Okumra, 1984; Lu and Wong, 1984). Other notable

symptoms reported included acneform lesions and hyperpigmentation. The incidence of hyperpigmentation was high (9/10) among offspring born to mothers pregnant during ingestion of the contaminated rice oil (Higuchi, 1976). This hyperpigmentation gradually faded over the 2-5 months after birth. Developmental tooth and bone defects were noted among some of the offspring (e.g. eruption of teeth at birth, larger-than-usual frontal and occipital fontanelles, and maintenance of a wider sagittal suture than usual). However, postnatal physical and mental development of infants born to contaminated rice oil-exposed mothers paralleled that of healthy infants (Higuchi, 1976).

While it was originally assumed that the rice oil was contaminated with only PCBs (Kanechlor 400), subsequent analysis revealed that there was also extensive contamination with polychlorodibenzofurans (PCDFs) and polychlorinated quaterphenyls (PCQs). Studies in animals have clearly indicated that PCDFs are more toxic than PCBs, and a number of lines of evidence support the contention that the adverse health effects suffered by the rice oil poisoning victims were a result of PCDF rather than PCB exposure. First, the concentrations of PCBs measured in patients with symptoms of Yusho were substantially lower than those observed in a number of studies of healthy, occupationally-exposed workers (Kunita et al, 1984; Masuda et al, 1985). The occupational studies have also shown workers to have undetectable or minimally-detectable concentrations of PCDFs (Kashimoto et al, 1985), while rice oil victims in Japan and Taiwan had significant concentrations of PCDFs, including a number of extremely toxic isomers. These human observations are supported by a study in which monkeys were fed PCBs with levels of PCDF and PCQ contamination similar to that of the Yusho oils (Kunita et al, 1984). Monkeys fed this diet developed dermatologic symptoms resembling those seen in patients with Yusho. Monkeys fed PCBs without PCDF/PCQ contamination, or fed PCQs alone, did not develop dermatologic lesions. Collectively these observations have led to the conclusion that the symptoms experienced by the rice oil poisoning victims were the result of PCDF rather than PCB exposure (Kuratsune, 1980; Kashimoto et al, 1981; Drill et al, 1982; Masuda et al, 1982; Masuda and Yoshimura, 1984; Chen et al, 1984; Kunita et al, 1985; Miyata et al, 1985, and Bandiera et al, 1984).

3.2 Health Effects Information from Environmental Exposure

A number of studies have attempted to measure health effects from environmental exposure to PCBs. The most common source of environmental exposure in these studies was the consumption of Lake Michigan fish (e.g. Humphrey, 1980; Fein et al, 1984; Kreiss et al, 1981; Smith, 1983), although the results from other sources of environmental exposure have also been studied (e.g. Baker et al, 1980). The results from these studies have generally been negative. Interpretation of the meaning of the few positive observations has been hampered by either the absence of a control population matched for known risk factors for the parameter studied, a strong positive correlation between PCB body burden and the body burden of other

halogenated organics, or the inability to demonstrate increased PCB body burden in the "exposed" population compared with background body burdens in "non-exposed" controls. The absence of these apparent adverse effects in studies of workers with much higher PCB exposures (see below) casts further doubt on the significance of these findings.

3.3 Health Effects Information from Occupational Exposure

With few exceptions, individuals with the largest and longest exposure to PCBs are found in the occupational setting. For this reason, the study of occupational exposure to PCBs is probably the best source of information regarding their health effects in humans.

A population of 326 workers employed at two capacitor manufacturing plants had been the most studied, and a number of reports concerning their health have appeared in the literature. Though this population had been exposed to Aroclor 1016 during the most recent two year period prior to the initial study (and to a lesser extent to Aroclor 1221), the long-term exposure was primarily to Aroclors 1242 and 1254 (Fischbein et al, 1979). Duration of exposure to PCBs was substantial, as 40% of the workers had been employed for 20 years or longer. Air levels of PCBs varied widely throughout the plants, ranging from 0.007 mg/m^3 to 11.0 mg/m^3 (Fischbein et al, 1982). Wolff et al (1982) analyzed 290 plasma samples and 61 adipose tissue samples from these workers and found plasma concentrations of lesser chlorinated PCBs ranging from 6-2350 ppb, plasma concentrations of higher chlorinated PCBs ranging from 0-546 ppb, adipose levels of lesser chlorinated PCBs of 0.6-414 ppm, and adipose levels of higher chlorinated PCBs from 1-165 ppm. Approximately half of the population had a history of dermatologic symptoms, rash being the most common. A history of non-adolescent acne, a symptom considered characteristic of PCB exposure, was reported by 12% of the workers (Fischbein et al, 1982). Edema of the upper eyelid, eye discharge, and enlargement of the Meibomian glands, common symptoms among patients with Yusho, were each found in 7% or less of the occupationally-exposed workers (Fischbein et al, 1985). Warshaw et al (1979) studied respiratory function in 243 of the workers and found that 14% had abnormal forced vital capacity. From an extensive examination of clinical chemistry parameters in the PCB-exposed worker population, Fischbein et al (1979) concluded :

"_ *there was a paucity of abnormal results in the biochemical studies. Similar findings were noted in the results of the hematologic tests* _ ".

Other occupationally-exposed populations have also been studied, and have been included in a recent review by Gaffey (1983) of the human health effects of PCBs. Gaffey, in surveying the human PCB literature, classified health effect observations into five categories: dermatologic findings, liver function, fat metabolism, blood and blood pressure, and symptoms, illnesses, and other

ıditions. Which respect to each of these categories, the ıllowing observations and conclusions were noted:

Dermatological effects: Of 11 studies of PCB-exposed workers which reported dermatologic findings, dermal symptoms were noted in all 11. Correlation of dermal symptoms with blood PCB concentrations were generally poor or non-existent. However, collectively the evidence strongly suggests that chloracne may occur when PCB blood levels exceed 150-200 µg/ml.

Liver function: Some abnormality in liver function indicated by a change in one or more relevant clinical chemistry parameters, was observed in five of seven studies of occupational exposure to PCBs. [Though not mentioned by Gaffey, it should be noted that while differences in some parameters indicative of liver function were observed to be statistically significant in some of these studies, these differences were uniformly quite small. Further, with chemical-induced hepatotoxicity one would expect to find a consistent pattern of abnormalities among overlapping indices of liver function. No such consistent pattern was observed.] In no case was the abnormality associated with any measurable adverse health effect. The remaining two studies found no evidence of liver abnormalities. A ninth study found evidence of induction of drug metabolism among PCB-exposed workers (Alvares et al, 1977).

Fat metabolism: There appears to be a correlation between serum triglycerides and PCB exposure in most studies. Results concerning cholesterol are equivocal, with one study showing an increase, one a decrease, and three no change. Conflicting results have also been observed with HDL-cholesterol. Changes in fat metabolism produced by PCBs, if they exist, do not appear to be of clinical significance.

Blood and blood pressure: None of the five studies which examined blood chemistry noted abnormalities associated with PCB exposure. One study of PCB-exposed workers measured blood pressure, but found no association with PCBs.

Symptoms, illness, and other conditions: Five studies report a variety of symptoms among PCB-exposed workers. Most of these symptoms appear to be unrelated to PCB exposure. None of the reports have found significant clinical effects to be associated with PCB exposure.

3.4 Summary of Non-cancer Human Health Effects of PCBs:

PCBs appear to be of low potency in producing adverse human health effects. Among workers with demonstrated high body burdens

of PCBs, the only consistently demonstrated clinical finding is dermatological abnormalities. Though one study has found evidence for hepatic enzyme induction in humans, there is no compelling evidence for PCB-induced liver injury despite relatively high levels of exposure. A number of subjective and objective symptoms have been reported for workers exposed to PCBs, and a variety of such symptoms would be expected in examining any population. The appearance of symptoms usually does not correlate with PCB levels or exposures, and no symptom or symptom type (other than dermal) is prominent when the studies are considered collectively. Exposure to PCBs highly contaminated with PCDFs may lead to significant symptomology, but these effects appear to be due to the more toxic PCDFs.

4.0 GENOTOXIC EFFECTS OF PCBS

4.1 Bacterial Mutagenicity Studies

Wyndham and co-workers (1976) were probably the first to test several chlorinated biphenyl mixtures in a bacterial test system. In this study the TA1538 mutant strain of *S. typhimurium* developed by Ames was used as the tester strain of bacteria, and rabbit liver microsomes were apparently added to metabolize the PCBs. Contrary to the authors' report that PCBs were weakly mutagenic under study conditions, a review of their data indicates that only 4-chlorobiphenyl and Aroclor 1221 demonstrated significant activity. Interestingly, Aroclor 1221, which has a chlorine content of 1.15 chlorine atoms per molecule and is therefore largely a monochlorobiphenyl mixture, was considerably less active than 4-chlorobiphenyl. A review of the actual results provided in the report indicate that neither 2,2',5,5'-tetrachlorobiphenyl or Aroclor 1268 are mutagenic.

Subsequent attempts to demonstrate that PCBs might cause mutations in bacterial test systems have failed. In an affadavit from Dr. Safe, senior author of the Wyndham et al (1976) article, he indicates, based on his own inability to reproduce these findings, that 4-chlorobiphenyl should not be considered mutagenic. Additional assays by other scientists also discount any suggestion that PCBs or monochlorobiphenyls are mutagenic in bacterial systems. Heddle and Bruce (1977; Bruce and Heddle, 1979) tested a number of different chemicals in the TA1535, TA1537, TA98, and TA100 strains of *S. typhimurium* , both with and without an S-9 mix to provide metabolism and possible activation of the compounds tested. Aroclor 1254 was not mutagenic in this study. Similarly McMahon et al. (1979) reported results after screening a large number of compounds in a testing scheme incorporating several auxotrophs of *S. typhimurium* and *E. coli* . The only chlorinated biphenyl tested was 4-chlorobiphenyl, but it was found to be negative, i.e. it was not mutagenic, in all 10 tester strains. Based on these studies and other reports in the literature (Schoeny et al., 1979; Rinkus and Legator, 1979; EPA, 1980;

TABLE 9

Summary of Microbial Mutagenicity Test Results

Product	Tester Strain	S. typhimurium										E. coli	
		C3076	D3052	G46	TA98	TA100	TA1000	TA1535	TA1536	TA1537	TA1538	WP2	WP2uvr
Aroclor 1268		-	-	-	-	-	-	-	-	-	Neg.	-	-
Aroclor 1254		Neg.	Neg.	Neg.	Neg.	Neg.	Neg.	Neg.	-	Neg.	Neg.	Neg.	Neg.
Kanechlor 500		-	-	-	Neg.	Neg.	-	-	-	-	-	Neg.	-
2,2',5-5-tetrachloro-biphenyl		-	-	-	Neg.	Neg.	-	Neg.	-	Neg.	Neg.	-	-
[illegible]nichlor		-	-	-	Neg.	Neg.	-	Neg.	Neg.	Neg.	Neg.	Neg.	-
Aroclor 1221		-	-	-	-	-	-	-	-	-	Neg.*	-	-
4-chloro-biphenyl		-	-	-	-	-	-	-	-	-	Neg.	-	-

Source: Levinskas (1981)
* Based on re-evaluation of Wyndham et al.(1975) by Dr. Safe.

Levinskas, 1981), all of which have been summarized in Table 9, it is concluded that PCBs are not mutagenic in bacterial test systems.

4.2 Clastogenic/Chromosomal Studies

Several test systems designed to measure chromosomal damage have been used to test the potential genotoxicity of PCBs. The only human cell line tested were human lymphocytes in the study of Hoopingarner et al. (1972). The test system used phytohemagglutin stimulated cells which were treated with 100 ppm Aroclor 1254 for the first 24 hours, then again during the last eight hours of the S and G_2 stages of the cell cycle and during the three hours of mitosis. Cytological examinations of the cells during these stages of the cell cycle failed to reveal any effect of PCB treatment on chromosomal integrity.

Green et al. (1975a) tested Aroclors 1242 and 1254 for their

ability to induce chromosomal damage in the bone marrow and sperm cells of rats. Aroclor 1242 was administered either in single doses of 1250, 2500 and 5000 mg/kg, or as four daily doses of 500 mg/kg; Aroclor 1254 was tested using three different dosage regimens, i.e. five daily doses of either 75, 150, or 300 mg/kg. The two highest doses of Aroclor 1254 and all doses of Aroclor 1242 caused a loss of body weight in these animals and the multiple doses of Aroclor 1242 killed half of the test animals. Even though considerable systemic toxicity was observed at these doses, neither Aroclor was found to induce chromosomal damage in sperm cells or bone marrow cells. Dikshith et al. (1975) similarly reported that PCBs do not produce chromosomal abnormalities in the sperm cells of rats exposed to seven daily dosages of 50 mg/kg of Aroclor 1254. The results of these two studies were later reproduced and substantiated by Garthoff et al. (1977). Male rats were fed dietary levels of 5, 50 or 500 ppm of Aroclor 1254 for 5 weeks after which sperm and bone marrow cells were examined for chromosomal damage; no evidence of chromosomal damage to either cell line was obtained.

Several investigators have employed the micronucleus test as a measure of chromosomal damage. Studies using this test system have reported that PCBs do not induce micronuclei in mice injected for five consecutive days with Aroclor 1254 at doses approximating one half of the LD_{50} or at doses representing some fraction thereof (Heddle and Bruce, 1977; Bruce and Heddle, 1979; Jenssen and Ramel, 1980). The doses used in these studies also failed to induce sperm abnormalities believed to be indicative of mutations or chromosomal deletions.

Using Drosophila melanogaster and Bombyx mori as the eukaryote test species, neither of the French PCB mixtures, Clophen 30 or Clophen 50 produced chromosomal aberrations (Nilsson and Ramel, 1974).

In conjunction with a study measuring the effects of PCBs on the breeding success of Ring Doves fed a 10-ppm diet of Aroclor 1254, Peakall et al. (1972) also reported the incidence of chromosomal aberrations in eggs of the birds. The average aberration rate changed from 0.8% in controls to 1.8% in the PCB treated birds. However, the average rate of chromosomal aberrations measured in the eggs of PCB pretreated birds was only higher than the highest control value in 4/17 eggs and the authors indicated that these results were inconclusive. By comparison, Aroclor 1242 has been injected into the eggs of white Leghorn chickens until PCB concentrations reached 20 ppm (Blazak and Marcun, 1975). Even though concentrations were clearly toxic, evidence of chromosomal aberrations was not observed.

4.3 Dominant Lethal tests

Green et al. (1975b) reported that Aroclor 1242 administered as single doses of 625, 1250 and 2500 mg/kg or after five daily dosages of 125 or 250 mg/kg failed to induce dominant lethal

mutations in the rat. Aroclor 1254 administered for five days at dosages of 75, 150 or 300 mg/kg or at dietary levels of 25 or 100 ppm for 70 days was likewise without effect. Keplinger et al. (1972) and Calandra et al. (1976) tested Aroclors 1242, 1254 and 1260 in mice at dosages of 500 or 1000 mg/kg and also found them to be without dominant lethal effects.

4.4 DNA Damage Studies

On the basis of sedimentation rates, Stadnicki et al. (1979) have reported that the epoxide of tetrachlorobiphenyl caused single-stranded chromosomal breaks in the DNA of L-929 cells at concentrations ranging from 1 ug/ml to 100 ug/ml. A mixture of two hydroxylated metabolites, and to a much lesser extent tetrachlorobiphenyl, caused some damage at 20 ug/ml and what was reported as significant damage at 100 ug/ml. The significance of this single *in vitro* test is questionable, given the fact that the epoxide metabolite was the only chemical species demonstrating a strong activity in this test system. This conclusion has been reached in part because the authors, on the basis of this study, similarly concluded that the epoxide metabolite is the only chemical species of interest with regards to its potential carcinogenicity. However, all mammalian tests as well those *in vitro* tests containing some activation system were negative, indicating that either the epoxide is not genotoxic in other test systems or quantities of the epoxide sufficient to produce genotoxicity are not generated *in vivo*.

4.5 Cell Transformation Studies

Norback et al. (1981) reported that Aroclor 1254 transformed C3H10T11/2 cells to Type III foci after six weeks of continuous exposure to 10 ug/ml of Aroclor 1254, while a concentration of 1 ug/ml did not. The authors suggested that these results indicate that the effects of PCBs in cell culture may include promotion. In contrast to this study, Pienta (1980) reported Aroclor 1254 did not induce cell transformations after eight days of exposure when utilizing Syrian hamster embryo cells. The highest dose tested was 50 µg/ml, which was five times the highest dose later used by Norback et al. (1981).

4.6 Summary

The results of the preceding studies have been summarized on the next page in Table 10. Given the fact that the only test reported as positive is one of questionable significance, the number of times PCBs have been tested and found to be without significant genotoxicity lead inevitably to the conclusion that these compounds should be considered to be without evidence of genotoxic activity.

Table 10

Species	Type of Genotoxic Activity: DNA damage	Mutation	Chromosomal damage	Cell Transformation
Prokaryotes		Neg.(28/12)	Neg.(2/2)	
Mammalian cells (*in vitro*)	Pos.?(1/1)		Neg.(6/9)	Neg. (2/2)
Mammals (*in vivo*)			Neg.(3/3)	
Human cells (*in vitro*)			Neg.(1/1)	

The first number in the parentheses indicates the total number of times an Aroclor was tested, the second number in parentheses indicates the total number of times a specific test strain or cell line was tested. Thus, the results followed by the larger numbers in parentheses represent the results most likely to be reproduced : further testing is performed in the future.

5.0 Summary of Data Reviewed and Evaluation

5.1 Evaluation of Animal Evidence for Carcinogenicity of PCBs

The investigation of PCBs' carcinogenic potential in mice is limited to two short studies, while some eight to ten studies have been reported using various strains of rat. The PCB mixtures tested thus far are : 1) in mice - Kanechlor 300, Kanechlor 400, Kanechlor 500 and Aroclor 1254; and 2) in rats - Kanechlor 300, Kanechlor 400, Kanechlor 500, Clophen A30, Clophen A60, Aroclor 1254 and Aroclor 1260.

PCBs have been found to be tumorigenic in mice with Aroclor 1254 producing hepatomas after 11 months of exposure and Kanechlor 500 (similar in composition to Aroclor 1254) inducing hepatocellular carcinomas after 8 months of exposure. These lesions were shown to be reversible and specific for the dose (500 ppm) and chlorination of the PCB mixture.

In rats, Aroclor 1260 or its equivalent, Clophen A60, have produced hepatocellular carcinomas in three studies at doses of approximately 100 ppm, a dose which appears to represent the maximally tolerated dose for rats. A review of these three studies indicates that the tumors occur very late in the life of the animal, with a significant incidence of tumors only beginning to appear after about two years of exposure. Of interest is the fact that in all three studies the PCB treatment, while increasing the

incidence of liver cancer, did not increase the total tumor incidence. The total tumor incidence was not increased in these studies because in each case the incidence of other tumor types had been significantly decreased. This suggestion of antitumor activity of PCBs has also been demonstrated in a study examining the effect of PCB exposure on the final tumor incidence in animals following the transplantation of the Walker 256 sarcoma. The effects of chronic PCB-treatment was not life-shortening, and in fact in two of the studies the morbidity and mortality of the animals was actually decreased by PCB treatment. Furthermore, while the tumors are described as malignant, i.e. hepatocellular carcinomas, in none of the three studies did the liver tumors metastasize to other organs even though metastases would be expected if the tumors were malignant. So from these studies it is evident that PCB-treatment does not increase the total cancer risk in these animals, rather it shifts the incidence of the type of tumors observed by significantly decreasing some tumor types while enhancing the incidence of liver tumors. Lastly, PCB mixtures of lesser chlorination, i.e. Aroclor 1254 and Clophen A30 (similar in composition to Aroclor 1242, see table 1.4, page 8, of Brinkman and DeKok, 1980), have been examined in two separate studies and found not to be carcinogenic. Thus, conclusions to be drawn from the rat data, like the mouse data, are specific for the dose and degree of chlorination of the PCB mixture being tested.

Possibly because PCBs produce liver hypertrophy, they enhance the tumorigenesis of certain liver carcinogens if given after the carcinogen in question has had an opportunity to initiate tumors. However, if the PCB exposure precedes or is concomitant with exposure to liver carcinogens the tumorigenic response is typically decreased, probably as a result of an enhanced metabolic detoxification of the carcinogen.

To summarize, the qualitative human relevance of the carcinogenic activity of PCBs based on the animal data is limited. The studies providing some evidence of its carcinogenic activity are specific for the degree of chlorination of the PCB mixture, the total tumor incidence is not increased, and the tumors produced occur only very late in the life of the animal and have no adverse effect on the morbidity or mortality of the animal. There are other considerations that limit concern for the carcinogenic data in rats as well. PCBs are not mutagenic, and the mechanism of tumorigenesis for these compounds therefore would appear to involve an epigenetic mechanism. There is also substantial evidence that the doses used to induce tumors in rats are hepatotoxic, and evidence indicating the neoplasms induced by PCBs are reversible if the exposure is terminated before the animal has been exposed for a considerable portion of the animal's lifespan. All of these findings seriously undermine the human relevance of the animal carcinogenicity data. This is particularly true as the human dosages from past and present human exposures are far lower than those used in the animal studies. Given these considerations, it is concluded that the animal data provides *sufficient evidence* of *limited human relevance* that PCBs of 60% chlorine content (e.g. Aroclor 1260/Clophen A60)

are carcinogenic in animals. For PCBs having a chlorine composition of 54% (e.g. Aroclor 1254/Kanechlor 500) there is only *inadequate evidence* of carcinogenicity in animals because the larger and longer study in rats was negative, a finding suggesting that the reversible effects reported in mice may have resulted from a promotion of the substantial background incidence of liver tumors occurring in this species. For the remaining commercial PCB mixtures (i.e. Aroclor 1248, Aroclor 1242, Kanechlor 400, Kanechlor 300, and Clophen A30) there is either *no evidence* or *insufficient evidence* for the carcinogenicity of these mixtures in animals.

5.2 Evaluation of Epidemiological Evidence for Carcinogenicity of PCBs

Evidence for human carcinogenicity would be supplied if the two major epidemiological studies of PCB-exposed workers demonstrated a consistent increase in cancer mortality. This increase should show a positive correlation with exposure and evidence of latency. The evidence for carcinogenicity would be strengthened if the incidence of specific types or sites of neoplasms were consistently elevated. In reviewing the data from these two cohorts, none of these conditions are met.

While Bertazzi et al. (1986), in their study of Italian PCB-exposed workers, found a statistically significant elevation in the rate of cancer mortality among both male and female workers, the much larger study of Brown found no increase in cancer mortality. In the Brown study, there were higher-than-expected incidences of rectal and liver cancer. Evidence for an association between PCB exposure and these malignancy types cannot be considered strong, however, in that: 1) no cases of rectal cancer were observed after the initial report, suggesting that this increased rate was anomalous, 2) the number of cases of liver cancer observed in this study is not appreciably greater than expected when examined without total number of liver neoplasms used by Brown which includes those liver cancers that have metastasized from other organs, 3) in comparison, the study by Bertazzi et al. found only one case of liver cancer and no cases of rectal cancer in their cohort, and 4) the absence of a clear association with latency or relationship with duration of exposure.

The primary sites of neoplasms contributing to the higher-than-expected cancer mortality rates in the Bertazzi studies were located in the digestive system and the hematopoietic and lymphatic systems. These were not increased in the larger cohort reported by Brown. Further, in the Bertazzi cohort, there was no evidence of latency or relationship between cancer mortality and exposure to PCBs. It should also be noted that in the larger of the two subdivisions of their cohort, the female workers, differences in incidences of causes of death presumably unrelated to PCB exposure (viz., increases in accidental death and decreases in deaths from cardiovascular disease) were of similar magnitude as increases in death from malignant tumors. This suggests, at least

r this group, that other confounding variables may exist.

In summary, epidemiological evidence for human carcinogenicity of PCBs is at present weak and mostly negative. As such, until larger epidemiological studies can be completed, the data must be considered *inadequate to characterize PCBs as human carcinogens*.

5.3 Evaluation

Aroclor 1260/Clophen A60 :

There is *sufficient evidence* of *limited human relevance* for the carcinogenicity of Aroclor 1260 in animals.

The human evidence for carcinogenicity of this compound is *inadequate* .

Aroclor 1254/Kanechlor 500 :

There is *inadequate evidence* for the carcinogenicity of Aroclor 1254 in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

Aroclor 1248/Kanechlor 400 :

There is *no evidence/insufficient evidence* for the carcinogenicity of Aroclor 1248 in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

Clophen A30/Kanechlor 300/Aroclor 1242 :

There is either *no evidence* or *insufficient evidence* for the carcinogenicity of these mixtures in animals.

The human evidence for carcinogenicity is negative but *inadequate*.

REFERENCES

Alvares, A.P. et al. 1977. Alterations in drug metabolism in workers exposed to polychlorinated biphenyls. Clin. Pharmacol. Ther. 22:140.

Anderson, L.M. et al. 1983. Effects of polychlorinated biphenyls on lung and liver tumors initiated in suckling mice by N-nirtosodimethylamine. J. Natl. Cancer Inst. 71:157.

Arai, M. et al. 1983. Comparative enhancing effects of polychlorinated biphenyls and phenobarbiyal on dimethylnitrosamine-induced hepatic and renal tumorigenesis in rats. In: Developments in the Science and Practice of Toxicology, A.W. Hayes, R.C. Schnell and T.S Miya (eds.), pp 359-362, Elsevier Science Publishers.

Bahn, A.K. et al. 1976. Melanoma after exposure to PCBs. New Engl. J. Med. 295:450.

Baker, E.L. et al. 1980. Metabolic consequences of exposure to polychlorinated biphenyls (PCB) in sewage sludge. Am. J. Epidemiology 112:553.

Bandiera, S. et al. 1984. Comparative toxicities of the polychlorinated dibenzofuran (PCDF) and biphenyl (PCB) mixture which persists in Yusho victims. Chemosphere 13(4):507.

Berry, D.L. et al. 1978. Lack of tumor-promoting ability of certain environmental chemicals in a two-stage mouse skin tumorigenesis assay. Res. Commun. Chem. Pathol. Pharmacol. 20:101.

Bertazzi, P.A. et al. In press. Cancer mortality of electrical workers exposed to PCB's. Am. J. Ind. Med.

Bertazzi, P.A. et al. 1981. Mortality study of male and female workers exposed to PCBs, presented at the International Symposium on Prevention of Occupational Cancer Helsinki, Finland.

Brinkman, U.A. Th. and A. DeKok. 1980. Production, properties and usage. In: Halogenated Biphenyls, Terphenyls, Naphthalenes, Dibenxodioxins and Related Products (ed. R.D. Kimbrough) Elsevier, New York, NY, p. 1-40.

Brown, D.P. 1986. Mortality of workers exposed to polychlorinated biphenyls - An Update. (under revision)

Brown, D.P. and M. Jones. 1981. Mortality and industrial hygiene study of workers exposed to polychlorinated biphenyls. Arch. Environ. Health. 36:120.

Calandra, J.C. 1976. Summary of toxicological studies on commercial PCBs. In: Proceedings of the National Conference on

Polychlorinated Biphenyls. USEPA report 560/6-75-004.

Chen, P.H.S. et al. 1984. Polychlorinated biphenyls, dibenzofurans, and quaterphenyls in the toxic rice-bran oil and PCBs in the blood of patients with PCB poisoning in Taiwan. Am. J. Ind. Med. 5:133.

Deml, E. et al. 1983. Benzo(a)pyrene initiates enzyme-altered islands in the liver of adult rats following single pretreatment and promotion with polychlorinated biphenyls. Cancer Lett. 19:301.

Deml, E. and D. Oesterle. 1987. Dose-reponse of promotion by polychlorinated biphenyls and chloroform in rat liver foci bioassay. Arch. Toxicol. 60:209.

DiGiovanni, J. et al. 1977. Tumor-initiating ability of 2,3,7,8-tetrachlorodibenzo-p-dioxin and Aroclor 1254 in the two stage system of mouse skin carcinogenesis. Bull. Environ. Contam. Toxicol. 18:552.

Drill, V.A. et al. 1982. Comments and Studies on the use of Polychlorinated Biphenyls in Response to an order of the United States Court of Appeals for the District of Columbia. Drill, Freiss, Hays, Loomis and Shaffer Inc., Consultants in Toxicology, Arlington, VA.

EPA (Environmental Protection Agency). 1980. Ambient Water Quality Criteria for Polychlorinated Biphenyls. EPA 440/5-80-068, PB81-117798.

Fein, G.G. et al. 1984. Prenatal exposure to polychlorinated biphenyls: Effects on birth size and gestational age. J. Ped. 105(2):315.

Fischbein, A. et al. 1979. Clinical findings among PCB-exposed capacitor workers. N.Y. Acad. Sci. 320:703.

Fischbein, A. et al. 1982. Dermatological findings in capacitor manufacturing workers exposed to dielectric fluids containing polychlorinated biphenyls (PCBs). Arch. Environ. Health 37:69.

Fischbein, A. et al. 1985. Oculodermatological findings in workers with occupational exposure to polychlorinated biphenyls (PCBs). Brit. J. Indust. Med. 42:426.

Gaffey, W.R. 1983. The epidemiology of PCBs. In: PCBS: Human and Environmental Hazards. F.M. D'ltri and M.A. Kamrin [Eds.], Butterworth Publ., Woburn, MA.

Gans, J.H. and S.J. Pintauro. 1986. Liver scarring induced by polychlorinated biphenyl administration to mice previously treated with diethylnitrosamine. Proc.Soc. Exp. Biol. Med. 183:207.

Greim, H. et al. 1985. Studies on the evaluation of tumor

promoting agents in human hepatocarcinogenesis., In: Hepatology : A Festschrift for Hans Popper, H. Brunner and H. Thaler (eds), Raven Press, New York.

Hayes, M.A. et al. 1985. Influence of cell proliferation on initiating activity of pure polychlorinated biphenyls and complex mixtures in resistant hepatocyte *in vivo* assays for carcinogenicity. J. Natl. Cancer Instit. 74:1037.

Hendricks, J.D. et al. 1980. Null effect of dietary Aroclor 1254 on hepatocellular carcinoma incidence in Rainbow trout (Salmo gairdneri) exposed to aflatoxin B_1 as embryos. J. Environ. Pathol. Toxicol. 4:9.

Hendricks, J.D. et al. 1977. Inhibitory effect of a polychlorinated biphenyl (Aroclor 1254) on aflatoxin B1 carcinogenesis in rainbow trout. J. Natl. Cancer Instit. 59:1545.

Higuchi, K. 1976. PCB Poisoning and Pollution, Academic Press, New York, N.Y.

Humphrey, H.E.B. 1980. Evaluation of humans exposed to halogenated biphenyls. Am. Chem. Soc. Div. Environ. Chem. Preprints 20(2):272.

Ito, N et al. 1978. Enhancing effect of chemicals on production of hyperplastic liver nodules induced by N-2-fluorenylacetamide in hepatectomized rats. Cancer Res. 41:3071.

Ito, N. et al. 1974. Histopathological studies on liver tumorigenesis in rats treated with polychlorinated biphenyls. Gann 65:545.

Ito, N. et al. 1973a. Histopathologic studies on liver tumorigenesis induced in mice by technical polychlorinated biphenyls and its promoting effects on liver tumors induced by benzene hexachloride. J. Natl. Cancer. Inst. 51:1637.

Ito, N. et al. 1973b. Interactions of liver tumorigenesis in mice treated with technical polychlorinated biphenyls (PCBs) and benzene hexachloride. In: New Methods in Environmental Chemistry and Toxicology, p. 141.

Kashimoto, T. et al. 1981. Role of polychlorinated dibenzofuran in Yusho (PCB poisoning). Arch. Environ. Health. 36:321.

Kashimoto, T. et al. 1985. PCBs, PCQs and PCDFs in blood of Yusho and Yu-Cheng patients. Environ. Health Persp. 59:73.

Kerklivet, N.I. and D.J. Kimeldorf. 1977a. Antitumor activity of a polychlorinated biphenyl mixture, Aroclor 1254, in rats innoculated with Walker 256 carcinosarcoma cells. J. Natl. Cancer Instit. 59:951.

Kerklivet, N.I. and D.J. Kimeldorf. 1977b. Inhibition of tumor

growth in rats by feeding a polychlorinated biphenyl, Aroclor 1254. Bull. Environ. Contam. Toxicol. 18:243.

Kimbrough, R.D. and R.E. Linder, 1974. Induction of adenofibrosis and hepatomas of the liver in Balb/cJ mice by polychlorinated biphenyls (Aroclor 1254). J. Natl. Cancer Institt. 53:547.

Kimbrough, R.D. et al. 1975. Induction of liver tumors in Sherman strain rats by polychlorinated biphenyl Aroclor 1260. J. Natl. Cancer Inst. 55:1453.

Kimura, N.T. et al. 1976. Polychlorinated biphenyls as a promoter in experimental hepatocarcinogenesis. Z. Krebsforsch. Klin. Onkol. 87:257.

Kimura, N.T. and T. Baba, 1973. Neoplastic changes in the rat liver induced by polychlorinated biphenyl. Gann 64:105.

Kreiss, K. et al. 1981. Association of blood pressure and polychlorinated biphenyls. J. Am. Med. Assoc. 245:2505.

Kunita, N. et al. 1984. Causal Agents of Yusho. Am. J. Ind. Med. 5:45.

Kunita, N. et al. 1985. Biological effect of PCBs, PCQs and PCDFs present in the oil causing Yusho and Yu-Cheng. Environ. Health Persp. 59:79.

Kuratsune, M. 1980. Yusho. In: Halogenated Biphenyls, Triphenyls, Napthalenes, Dibenzodioxins and Related Products, R. Kimbrough (ed.), Elsevier/North Holland, New York, NY, p. 287.

Levinskas, G. 1981. "A review and evaluation of carcinogenicity studies in mice and rats and mutagenicity studies with polychlorinated biphenyls." Monsanto publications.

Loury, D.J. and J.L. Byard. 1983. Aroclor 1254 pretreatment enhances the DNA repair response to amino acid pyrolysate mutagens in primary cultures of rat hepatocytes. Cancer Lett. 20:283.

Lu, Y.C. and P.N. Wong. 1984. Dermatological, medical, and laboratory findings of patients in Taiwan and their treatments. Am. J. Ind. Med. 5:81.

Makiura, S. et al. 1974. Inhibitory effect polychlorinated biphenyls on liver tumorigenesis in rats treated with 3'-methyl-4-dimethylaminoazobenezene, N-2-fluorenylacetamide and diethylnitrosamine. J. Natl. Cancer Instit. 53:1253.

Masuda, Y. et al. 1982. Comparison of causal agents in Taiwan and Fukuoda PCB poisonings. Chemosphere 11:199.

Masuda, Y. and H. Yoshimura. 1984. Polychlorinated biphenyls and dibenzofurans in patients with Yusho and their toxicological

significance. Am. J. Ind. Med. 5:31.

Masuda, Y. et al. 1985. PCB and PCDF congeners in the blood and tissues of Yusho and Yu-Cheng patients. Environ. Health Persp. 59:53.

Miyata, H. et al. 1985. PCBs, PCQs and PCDFs in tissues of Yusho and Yu-Cheng patients. Environ. Health Perspect. 59:67.

Morgan, R.W. et al. 1981. Aroclor 1254-induced intestinal metaplasia and adenocarcinoma in the glandular stomach of F344 rats. Cancer Res. 41:5052.

Nagasaki, H. et al. 1972. Hepatocarcinogenicity of polychlorinated biphenyls in mice. Gann 63:805.

NCI (National Cancer Institute). 1978. Bioassay of Aroclor 1254 for Possible Carcinogenicity. DHEW publication No. (NIH) 78-838.

Nishizumi, M. 1980. Reduction of diethylnitrosamine-induced hepatoma in rats exposed to polychlorinated biphenyls through their dams. Gann 71:910.

Nishizumi, M. 1976. Enhancement of diethylnitrosamine hepatocarcinogenesis in rats by exposure to polychlorinated biphenyls or phenobarbital. Cancer Lett. 2:11.

Norback, D.H. and R.H. Weltman. 1985. Polychlorinated biphenyl induction of hepatocellular carcinoma in the Sprague-Dawley rat. Environ. Health Persp. 60:97.

Okumura, M. 1984. Past and current medical states of Yusho patients. In: Progress in Clinical and Biological Research, Vol. 137, Alan R. Liss, Inc., New York.

Osterle, D. and E. Deml. 1984. Dose-dependent promoting effect of polychlorinated biphenyls on enzyme-altered islands in liver of adult and weanling rats. Carcinogenesis 5:351.

Pereira, M.A. et al. 1982. Promotion by polychlorinated biphenyls of enzyme-altered foci in rat liver. Cancer Lett. 15:185.

Preston, B.D. et al. 1981. Promoting effects of polychlorinated biphenyls (Aroclor 1254) and dibenzofuran-free Aroclor 1254 on diethylnitrosoamine-induced tumorigenesis in the rat. J. Natl. Cancer Instit. 66:509.

Schaeffer, E. et al. 1984. Pathology of chronic polychlorinated biphenyl (PCB) feeding in rats. Toxicol. Appl. Pharmacol. 75:278.

Stott, W.T. and R.O. Sinnuber. 1978. Trout hepatic enzyme activation of aflatoxin B1 in a mutagen assay system and the inhibitory effect of PCBs. Bull. Environ. Contam. Toxicol. 19:35.

Shelton, D.W. et al. 1984a. Effect of dose on the inhibition of carcinogenesis/mutagenesis by Aroclor 1254 in Rainbow trout fed aflatoxin B1. J. Toxicol. Environ. Health 13:649.

Shelton, D.W. et al. 1984b. The hepatocarcinogenicity of diethylnitrosamine to Rainbow trout and its enhancement by Atoclors 1242 and 1254. Toxicol. Lett. 22:27.

Ward, J.M. 1985. Proliferative lesions of the glandular stomach and liver in F344 rats fed diets containing Aroclor 1254. Environ. Health Persp. 60:89.

Warshaw, R. et al. 1979. Decrease in vital capacity in PCB-exposed workers in a capacitor manufacturing facility. N.Y. Acad. Sci. 320:277.

Wolff, M.S. 1982. Body burden of polychlorinated biphenyls among persons employed in capacitor manufacturing. Int. Arch. Occup. Environ. Health. 49:199.

Young, S.S. 1985. Letter to the Editor. Toxicol. Appl. Pharmcol. 78:321.

LIST OF TABLES

EXPOSURE ASSESSMENT FOR SMALL BURROWING ANIMALS

Daily intake of PCBs (mg/kg/day) (I)

PCBs are taken in through the skin, the lungs, and the mouth.

$I = I_D + I_I + I_O$

I_D = Dermal intake
I_I = Inhalation intake
I_O = Oral intake

Dermal uptake is a function of the efficiency of absorption through the skin, the surface area exposed, and the PCBs in the soil covering the skin.

$I_D = E_D S L C_S$

$= 0.05/day \times \frac{36cm^2}{0.03kg} \times 10^{-6}kg/cm^2 \times C_S$

$= 0.00006/day\ C_S$

$= 0.00006/day\ C_S$

E_D = Efficiency of absorption of PCBs in soil through the skin
= 0.05/day (Hwang et al 1986 p.12-16)

S = Surface area exposed on a 30g vole

= 36 cm^2/30g/day. Includes ear pinnae, legs, underbelly, face and tail

L = Soil layer, assumed = 1.0 mg/cm^2
= $10^{-6}kg/cm^2$

C_S = Concentration in the soil

NOTE: There are no published reports of dermal uptake of PCBs in soil for voles. The rate chosen was one assumed for PCBs by Hwang (1986). Poiger and Schlatter (1979) report an uptake rate of 2.2%/day for TCDD in rats. The skin of rats is thicker than the skin of most voles, so absorption would be expected to be somewhat higher in voles.

The soil layer on the skin of the vole was estimated using the assumption of two times the soil covering found by Lepow et al (1975) on urban children (2 x 0.5116 mg/cm^2 = 1.023). This may be an underestimate for ear pinnae and other areas that are relatively naked and well vascularized but an overestimate in furry areas.

Inhalation intake

The concentration of PCBs in the vapor of the burrow was calculated using the method of MacKay (1985 and pers. comm.) assuming equal fugacities in the soil and air.

Concentration of PCBs in the air of a burrow

$K_{AW} = H/RT$

$K_{AW} = \frac{H\ (Pa\ m^3/mol)}{8.314\ Pa\ m^3/mol\ K^\circ \times T}$

$= \frac{202.6\ Pa\ m^3/mol}{(8.314\ Pa\ m^3/mol\ K^\circ \times 283.2\ K^\circ}$

Source: Mackay 1985

K_{AW} = Air water partition coefficient

T = Temperature assumed 10°C
= 283.2K°

R = 8.314 Pa m^3/mol K

H = Henry's law constant

= 2.0 x 10^{-3} atm m^3/mol 25°C (Aroclor 1254)

= 2.0 x 10^{-3} atm m^3/mol x $\frac{101325\ Pa}{atm}$

= 202.6 Pa m^3/mol

$K_{AW} = 0.0860$

$K_D = K_{OC}OC$

= 45392 x 0.02
= 907.84

K_D = Soil water partition coefficient (L/kg)

OC = Fraction organic carbon (assumed = 0.02)

K_{OC} = Soil organic carbon partitioning coefficient (following Kenaga and Goring 1980) (log K_{OC} = 0.544 log K_{OW} + 1.377)

p = Soil bulk density (assumed to be = 1.04 g/cm^3, the average in Brady (1974) (p. 53) for uncropped soil)

$K_{AS} = K_{AW}/pK_D$

= 0.0860/(907.84 x .96 L/kg)

= 9.868 x 10^{-5}kg/L

K_{AS} = Air soil partitioning coefficient

Bush et al (1986) measured the concentration in air over contaminated soil. When these data are converted to an estimate of the K_{AS}, they yield a value of 1.186 x 10^{-6}kg/L. As expected, this is below the theoretical constant at equilibrium. Even on the calm day that the measurement was taken, one can expect that equilibrium will not be attained because of some diffusion and convection away from the site.

As a worst case, consider small mammals like the pocket gopher, or mole, which spend virtually all of their time in burrows, with the burrow sealed off (Chase et al 1983); (Yates and Pedersen 1983).

Inhalation is assumed to have two phases, an active phase and a resting phase. The animal is assumed to have the same activity patterns as a pocket gopher (Chapman and Feldhamer, 1982 p. 248) and a respiration rate ratio (active: resting) equivalent to that given by Vleck (1979) for pocket gophers. It is assumed that vapor and dust are inhaled during the active phase, but only vapor is inhaled during the resting phase.

$I_I = C_S (K_{AS} (V_A + V_R) + V_A D)/W$

$= C_S (K_{AS} (RT_A M + RT_R) + RT_A MD)/W$

$= C_S (K_{AS} R (T_A M + T_R) + RTAMD)/W$

$= C_S$ (9.868 x 10^{-5}kg/L x 1.5 L/hr (9 hr/day x 4.1 + 15 hr/day) + (1.5 L/hr x 9 hr/day x 4.1 x 10^{-10}kg/L)/0.03kg

$= C_S$ x 0.256/day

V_A = Volume inhaled during active part of day = $RT_A M$ (L/day)

V_R = Volume inhaled while resting = RT_R (L/day)

R = Resting respiration rate = 1.5 L/hr

T_A = Time spent active per day = 9 hr/day

M = Active/resting rate = 4.1

T_R = Time spent resting/day = 15 hr/day

D = Dust level (assumed = 10 mg/m^3 USEPA) = 10^{-10}kg/L

W = weight = 30g = 0.03kg

Oral intake

Oral intake is a function of the ingestion rate of food, soil, and water, the concentration of PCBs in plants as a function of the concentration in soil, and the efficiency of dietary absorption.

$I_O = C_S(I_W + I_F + I_S)$

$I_W = R_W C_S/K_D$

$= 0.01\ C_S/907.84$

$= 1.1 \times 10^{-5} C_S$

I_W = Ingestion rate of PCBs in water

I_F = Ingestion rate of PCBs in food

I_S = Ingestion rate of PCBs in soil assumed to be 10% of diet

R_W = Rate of ingestion of water = 10%/day

R_F = Rate of ingestion of food = 0.2/day

$I_F = R_F BCF_P C_S$

$= 0.2/day \times 0.10\ C_S$

$= 0.02/day\ C_S$

BCF_P = Bioconcentration factor for plants

= 10% (Bush et al 1986)

K_D = Soil water partitioning coefficient

$C_W = C_S/K_D$

$I_S = R_S\ C_S$

$= .01\ C_S$

R_S = Rate of ingestion of soil (assuming 5% of diet is food)

$= .05 \times R_F = .01$

$I_O = C_S\ (1.1 \times 10^{-5}/day + 0.02/day + 0.01/day)$

$= 0.030/day\ C_S$

$I = I_D + I_I + I_O$

$= C_S\ (0.00006/day + 0.256/day + 0.030/day) \times C_S$

$= 0.286/day \times C_S$

A diet of 1 ppm PCBs has been found to increase the liver weights in F_1 male weanling rats (Linder et al 1974) and decrease the circulating levels of adrenal cortex hormone B (Byrne et al 1988). If a small mammal consumes 20 percent of its body weight per day, 1 ppm in the diet is equivalent to 0.2 mg/kg/day. At this dietary level, the soil concentration would be 0.7 mg/kg.

if 1 ppm => 1 mg/kg in food $\times \frac{0.2\ kg\ food}{kg\ body\ wt/day}$ = 0.2 mg/kg/day

if 10 ppm => 1 mg/kg $\times \frac{0.2\ kg\ food}{kg\ body\ wt/day}$ = 2 mg/kg/day

$0.2\ mg/kg/day = (0.286/day)\ C_S$

$C_S = \frac{0.2\ mg/kg/day}{0.286/day} = C_S$

$= 0.699\ mg/kg$

A diet of 10 to 20 ppm decreases the weight of reproductive organs, growth rate and reproductive success of second generation white-footed mice (Linzey 1988, Linder 1974). At this dietary level, the soil concentration would be 7 mg/kg to 14 mg/kg.

$$C_s = \frac{2 \text{ mg/kg/day}}{0.286/\text{day}}$$

$$= 6.99 \text{ mg/kg}$$